As described in Chapter 3, the negative effects of elevated blood lead levels on human health are well documented in the scientific literature. Additionally, lead is considered a non-threshold toxin meaning no level of lead exposure is considered safe (ATSDR, 2007). In 1985, EPA considered providing a reference dose for inorganic lead and determined it was inappropriate given that "changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold" (U.S. EPA, 2004a). Depending on the chemical make-up and physical properties of lead, it can be ingested or inhaled and once exposure occurs it may impact multiple organ systems. The proposed rule to ban the manufacture of lead wheel weights will benefit human health by reducing environmental exposures to lead.

Based on EPA's 2006 Lead Air Quality Criteria document (AQCD) and the available literature on the health effects of lead in humans and rodents, Section **Error! Reference source not found.** of this report provides a summary of the key health endpoints. Overall, the adverse impacts caused by lead contamination on mammalian health are well documented. To date, there is no level of lead exposure that has been identified, with confidence, as being clearly not associated with possible risk of deleterious health effects (U.S. EPA, 2006). The health effects associated with lead exposure can be grouped into four main endpoint categories as described by U.S. EPA (2006): (1) neurological effects (including IQ-related impacts) in children; (2) other health effects in children; (3) health effects in adults; and (4) toxicological evidence of health effects in animals. Each of these endpoints is discussed in detail in Section **Error! Reference source not found.** through **Error! Reference source not found.**.

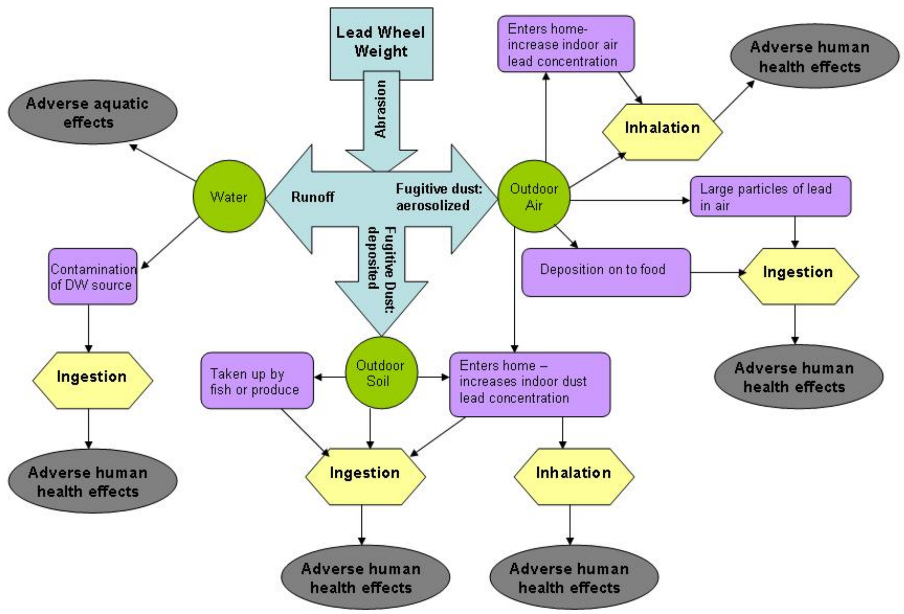
This chapter will present both a qualitative and quantitative discussion of the benefits of the proposed rule. Section 1.1 presents the qualitative discussion of benefits for this rule by describing the various routes of exposure that may exist with lead wheel weights. Section 1.2 presents five case-study scenarios used to quantify the societal benefits from eliminating lead wheel weights as a source of exposure to lead. Section **Error! Reference source not found.** summarizes the results of the benefits analysis.

1.1 Lead Exposure from Wheel Weights

According to Root (2000), lead loading of urban streets by motor vehicle weights is continuous, significant, widespread and potentially a major source of human lead exposure. In addition, adverse effects of lead are seen at a level so low that lead is one of the few noncarcinogens considered to be essentially without a threshold by EPA (2004a). Thus, reducing the amount of lead entering the environment by any means will result in some level of human health benefits. Given that lead may remain in the environment for anywhere between two to 2,000 years, the continuing release of lead from wheel weights has the potential to adversely affect human health and the environment for many future generations (Watmough, 2005 as cited in U.S. EPA, 2005b; Benninger, 1977 as cited in U.S. EPA, 2001a).

Figure 2-1 gives a summary of how lead in wheel weights may be dispersed into the environment and the routes by which people may be exposed to lead from wheel weights. The green circles represent the primary media of exposure; these media are not independent of one another and thus are connected by the blue arrows. For example, lead that enters soil from deposited dust may enter a waterway after a rain event through runoff. The purple rectangles display the secondary routes by which an individual can be exposed to lead from wheel weights. The yellow hexagons represent the means by which lead from wheel weights can enter the body and the gray ovals show the adverse effects associated with the exposure. Sections 1.1.1 and 1.1.2 discuss the multiple routes by which humans and the environment are exposed to lead from wheel weights and the associated adverse human health impacts.

Figure 2-1: Routes of Exposure and Impacts from Doses of Lead from Wheel Weights



1.1.1 Distribution of Lead from Wheel Weights into the Environment

Wheel weights are composed of 95 percent lead and 5 percent antimony. These weights are known to frequently fall off cars and can be broken down and dispersed by traffic so that half of the lead deposited in the street is no longer visible in little more than one week (Root, 2000). The amount of lead that enters the environment from wheel weights is dependent upon the number of vehicles on a given road, traffic patterns, vehicle weight, speed and weather. The more cars there are on a road, the higher the probability that lead from wheel weights will enter the environment. Additionally, the likelihood that wheel weights fall off increases in stop-and-start traffic (U.S. EPA, 2005b). The weight of the vehicles on the road also impacts the break-down and dispersion rate of the wheel weights by mechanically grinding the wheel weight into smaller particles and creating turbulence which ejects lead particles from the road.

Once the lead enters the environment, according to the CDC's Agency for Toxic Substances and Disease Registry (ATSDR, 2007), "lead is transferred continuously between air, water and soil by natural chemical and physical processes such as weathering, runoff, precipitation, dry deposition of dust and stream/river flow". Lead compounds may be transformed in the environment to other lead compounds; however lead is an element and thus cannot be destroyed. Therefore, once lead from wheel weights enters the environment, the potential for human and environmental exposures continues to exist for many generations. Figure 2-1 shows how lead from wheel weights has the potential to enter all three environmental media. The fate and transport of lead as it relates to air, water and soil is discussed in Sections 1.1.1.1, 1.1.1.2, and 1.1.1.3, respectively.

1.1.1.1 Air

The process by which lead from a wheel weight enters the environment begins with the wheel weight falling off of a vehicle and being deposited on a section of the road that is exposed to traffic (Root 2000). The weight can then be broken down into finer particles and released into the air by either natural wind or turbulence created by cars on the road. This leads to the initial dispersal of lead from a wheel weight beyond the road environment.

The fate of lead once it enters the air is highly dependant on the lead particle's size. Small particles can travel thousands of kilometers in the air and have a residence time, in air, up to ten days. However, coarse particles (those greater than 2 µm in diameter) will deposit more rapidly and closer to the source of the emissions. Lead deposition will occur either due to gravitational settling or wet deposition (ATSDR, 2007). It is highly likely the particles formed from lead wheel weights will deposit rapidly and close to the roadway. This assumption is made with the knowledge that the lead particles from the wheel weights are created by a mechanical process, traffic, which leads to coarse particle formation. When lead deposits, it has the potential to enter either soil or water depending upon the location of the deposition.

1.1.1.2 Water

Once the weight falls off a car, it has the potential to enter a water body by runoff from the road into a storm drain, runoff from soil into a storm drain, runoff directly from road or soil to a water body, percolation through soil into groundwater, or deposition from air into a water body. The route by which the lead can enter a water body is dependent upon the location in which the wheel weight is deposited. For example, not all cities have storm sewer systems which directly connect to a waterway; some are designed to siphon all water through a water treatment plant before it enters a body of water. Therefore, if the lead wheel weight falls off an area with this type of sewer system it is possible for the lead to reside in the water treatment plant without entering a body of water (Ustun, 2009). However, if a city does have a storm sewer system that connects directly to a body of water, the lead can directly enter the water body to which the sewer system is connected.

Lead has a tendency to bind tightly with soil and sediment and therefore does not often percolate into groundwater from soil. For the same reason, lead will not easily be transported in runoff from soil into a waterway. Therefore, the runoff of lead from a yard or other soil-based surface into a body of water will only occur in areas that are adjacent to a water body when soil may erode into the aqueous environment (ATSDR, 2007). In highly acidic conditions, when lead has to compete for binding sites on the soil with hydrogen ions there is a higher probability of free lead available to percolate into the groundwater or runoff into nearby waterways.

Once lead enters a water body, it is unlikely to dissolve in the water, but will instead bind to suspended solids or sediment. Lead has a tendency to bind to particles in both fresh and saltwater systems.

Given the behavior of lead in aquatic environments, lead from wheel weights will likely bind to particles or sediment in a water body rather than dissolve into the aqueous environment. According to ATSDR (2007), the ratio of lead in suspended solids to lead in dissolved forms varies from 4:1 in rural streams to 27:1 in urban streams. Very little lead is found in lakes, rivers or groundwater used to supply the public with drinking water. However, lead is toxic to all aquatic biota and those higher on the food chain may experience lead poisoning as a result of eating lead contaminated food (ATSDR, 2007).

1.1.1.3 Soil

After a wheel weight falls off a vehicle and is ground into finer particles by traffic and ejected from the roadway due to either natural wind or wind created by traffic, it can be deposited on soil. Soil is a significant media for exposure to lead and particles distributed into gardens and homes by the wind are believed to be the most significant source of exposure via inhalation or ingestion (Clark et al., 2006). Once lead enters soil, its mobility and bioavailability are largely controlled by the chemical and minerologic make-up of the soil. However, it is likely that lead will strongly adsorb to soil and because of this, lead in soil is usually retained in the upper layers and does not leach appreciably into the subsoil or groundwater, as mentioned in Section 1.1.1.2. Additionally, according to ATSDR (2007), the accumulation of lead in most soils is primarily a function of the rate of deposition from the atmosphere, and very little is transported through runoff to surface water or leaching to groundwater except under acidic conditions or when lead-containing soil particulates are more likely to erode into surface water.

Once lead enters a soil environment, the amount of time it will stay in the soil (its residence time) is also highly variable. Estimates for the residence time of lead vary widely, ranging from approximately two years to approximately 2,000 years. Therefore, depending on the environment in which lead is deposited it could result in potential exposure for tens to thousands of years (Watmough, 2005 as cited in U.S. EPA 2005b; Benninger, 1977 as cited in U.S. EPA 2001a).

1.1.1.4 Additional Considerations

In addition, the fate of lead being highly dependent upon the physiochemical properties of the location in which it is deposited, the meteorology of an area will also affect how and where lead from wheel weights enters the environment. For example, if the area of deposition is arid, similar to the southwestern U.S., water contamination may not be a large concern, but air or soil contamination may be. On the contrary, if the area has a large amount of rainfall, such as the Pacific Northwest, runoff may be a bigger concern. Furthermore, day to day meteorology can impact lead dispersion. Given changes in wind speeds and direction, the same locations may be differently impacted on any given day due to the instability of meterologic conditions.

Although quantitative estimates of the impact of lead from wheel weights in each environmental media are not presented here, a ban of lead wheel weights will decrease the amount of lead entering the environment into all media types.

1.1.2 Human Exposure and Dosing of Lead from Wheel Weights

This section describes the potential routes of exposure to the lead released into the environment from lead wheel weights and the factors that make exposure to lead extremely variable between individuals. The main routes of lead exposure are inhalation and ingestion. Another possible route, dermal exposure, is very inefficient and unlikely to occur (ATSDR, 2007).

Section 1.1.1.2 presented information about lead's behavior in an aquatic environment. Given that little lead is found in lakes, rivers or groundwater used to supply public drinking water systems, this is not a media of concern when considering how individuals are exposed to lead from wheel weights. Despite this, it is necessary to note that lead from wheel weights will still continuously be added to the aquatic environments and may adversely affect the aquatic ecosystem in which it is deposited.

Individuals are likely exposed to lead from wheel weights through ingestion of soil, food on which lead has deposited, or food grown in contaminated soil and dust; and through inhalation of air particles containing lead or dust containing lead. The amount of lead a person is exposed to that enters an individual's body is highly variable and dependent on a multitude of factors. The factors affecting dose by inhalation and ingestion are presented in section 1.1.2.1 and 1.1.2.2, respectively. However, it is important to note that although the exposure and dose an individual may experience varies greatly, lead is considered a non-threshold pollutant and therefore any dose has the potential to be detrimental to an individual's health.

1.1.2.1 Inhalation

Exposure through the inhalation route may occur when lead from wheel weights is aerosolized and the lead-containing dust is ejected from the roadway. Individual exposure is highly variable and dependent on an individual's behavior, age, and location of exposure.

Personal behaviors can affect a person's breathing rate and consequently the amount of lead the individual inhales. For example, if an individual is exercising, his or her breathing rate is likely higher than average. Therefore, an exercising person will inhale more lead from wheel weights than an individual breathing at an average rate.

Additionally, age related factors such as airway geometry and air-stream velocity within the respiratory tract will mediate the fraction of lead absorbed through the respiratory track: the greater the surface area of the airway, the more likely a particle will be absorbed. Additionally, particle size impacts the rate of absorption. Particles that are smaller than 1µm can be almost completely absorbed by the respiratory tract. However, larger particles will likely be caught and transferred by the mucocillary system and then sent to the esophagus and swallowed (ATSDR, 2007). Exposure by ingestion is discussed in section 1.1.2.2.

Location can also impact lead exposure. If an individual lives on a highly trafficked roadway, he or she has a higher probably of being exposed to lead from wheel weights, given more vehicles on the road.

In addition to exposure to lead from outdoor air, lead exposure in the indoor environment is also possible. Lead can travel into a home by air or can be tracked in by soil on shoes or clothing. The higher the lead concentration is in the outdoor environment, the higher the lead concentration will be indoors. Therefore, if an area has high outdoor lead levels due to lead wheel weights, the indoor lead concentration will also be high (U.S. EPA, 2008a). If lead becomes airborne in dust or as small particles, an individual may inhale it and ultimately absorb the lead into his or her blood.

Exposure to indoor and outdoor lead can lead to serious health effects as discussed in Chapter **Error! Reference source not found.**, especially for children. According to the California Environmental Protection Agency, children are more susceptible to the effects of lead and other air pollutants because their immune systems and developing organs are still immature (OEHHA, 2003). Banning lead wheel weights would reduce future lead emissions to the environment and thus would decrease subsequent symptoms associated with lead inhalation exposure. Furthermore, because lead that enters the environment has the potential to adversely impact people for thousands of years (Benninger, 1977 as cited in U.S. EPA, 2001a), a ban on lead wheel weight may prevent a great number of long term impacts associated with the inhalation of lead.

1.1.2.2 Ingestion

Lead can be ingested in the form of soil, indoor dust, food or coarse particles that were captured, after inhalation, by the mucocillary pathway and transferred to the esophagus and consequently swallowed. As with the indoor air lead concentration, indoor dust lead concentration is positively correlated with outdoor soil and air concentrations. Therefore, the greater the lead concentration in outdoor soil and air, the greater the concentration of lead in indoor dust to which people can be exposed to in the home. Lead exposure and subsequent dose due to each of these media (soil, dust, food, and large aerosolized particles), is highly variable and dependent on the physiology of the individual and the particle ingested. This is especially important with children because compared with adults; a larger proportion of the amount of lead ingested will enter the blood stream in children (ATSDR, 2007). The efficiency of gastrointestinal absorption of lead in food and beverages in children has been estimated to be around 40 percent (CDC 1991).

Lead is commonly found in soil, especially near roadways (ATSDR, 2007). This is especially important for children, who are still partaking in hand-to-mouth behavior and are more likely to ingest soil. LaGoy (1987) concluded, on average, children ingest twice as much soil per day than adults. Additionally, lead absorption after ingestion is higher in infants and children (two weeks to eight years old) than in adults. It has been shown that children absorb 40 to 50 percent of lead that is ingested (Alexander et al., 1974; Ziegler et al., 1978, both as cited in ATSDR, 2007) compared to adults who absorb only 3 to 10 percent of ingested lead (Heard and Chamberlain, 1982; James et al., 1985; Rabinowitz et al., 1980; Watson et al., 1986 all as cited in ATSDR, 2007).

An individual's nutritional status also attenuates the absorption of lead. Individuals that have food present in their digestive system absorb less lead. Additionally, iron and calcium deficiency both are inversely correlated with lead uptake. In other words, individuals that are deficient in these nutrients will absorb a higher amount of lead than those who are replete in these nutrients. Lastly, absorption of lead also increases during pregnancy (ATSDR, 2007).

If an individual ingests lead that entered the environment from a wheel weight, especially if he or she is a malnourished child or a pregnant woman, he or she is more likely to experience the adverse health effects outlined in Chapter 3. Although variable, ingestion is a common exposure route and lead wheel weights will continue to contribute to the amount of lead in soil, air and on and in food without a ban. Therefore, prohibiting the distribution of lead wheel weights would decrease the amount of lead in the environment and subsequently decrease future generations' probability of ingesting lead and with it, the probability of adverse health affects associated with lead.

1.1.2.3 Additional Considerations

When assessing the impact of lead from wheel weights, background levels should be considered. When lead wheel weights are deposited in an area with new development,(i.e. homes without a history of lead paint and soil that has not been exposed to leaded gasoline emissions), lead from wheel weights contribute a large proportion of an individual's lead exposure. On the other hand, when lead wheel weights are deposited in an area with high background lead concentrations, (i.e. a historic urban area with old homes that may have had lead paint), the lead from the wheel weights may contribute only a small proportion of the lead to which people are exposed. The population living in these areas is likely to be exposed to lead from other sources. Additionally, for each exposure and dosing scenario, one must consider that the exposure and subsequent health effects may not be immediate. When lead enters the environment it can stay for many generations and therefore the full extent of the impacts may not be seen for possibly ten or hundreds of years into the future.

1.2 Case Studies: Near-Roadway Residence Exposure Scenarios

EPA conducted five case studies in order to quantify a subset of the benefits from banning lead wheel weights. The approach for these case studies is to determine the IQ decrement that children experience from exposure to lead from wheel weights in near-roadway residences. As a simplifying assumption, individuals are not assumed to experience any reduced exposures due to the lead wheel weight ban until seven years after the ban, which is when nearly all lead wheel weights are expected to be eliminated from use (see 1.2.4). The general approach is to estimate the monetary benefits of avoiding the IQ decrement associated with exposure to lead from wheel weights for a subset of the U.S. population under the age of seven that lives in residences near roadways.

Five case studies were conducted, each in a unique hypothetical location. The hypothetical locations chosen for the case-studies represents a wide range of locations in which people live in the United States and subsequently, a wide range of areas in which benefits from this rule will be realized. In each case study, one child's IQ decrement in a single residence was determined. Specifically, EPA examined a child's exposure to lead wheel weights from age zero to seven years from inhalation and ingestion of contaminated soil and indoor dust and translated the lead exposures into the child's blood lead concentration and resulting IQ change. EPA recognizes that there are other routes of exposure not included in this case study, for example ingestion of drinking water contaminated with lead from wheel weights. However, EPA believes that the exposure scenarios included in the case studies, i.e., inhalation of outdoor and indoor air and ingestion of soil and dust, encompass the majority of lead exposure that can be attributed to wheel weights.

The subsequent sub-sections lay out the exposure modeling inputs and steps, and the process used to quantify the benefits from the exposure model results. Section 1.2.1 explains in detail each hypothetical residential area for which a case study was conducted and Section 1.2.2 explains the steps, inputs and assumptions of the exposure model. In Section 1.2.3, the process of translating the modeled exposure concentrations to blood lead concentrations and the resulting IQ changes is described. Given that the proposed rule would only ban the use of lead for new wheel weights, the benefits will not be seen for several years due to vehicles that still carry existing lead wheel weights; therefore, Section 1.2.4 explains the process EPA used to determine the point in time at which the benefits will begin to be seen. The monetization of benefits is then presented in Section **Error! Reference source not found.**. Lastly, in section **Error! Reference source not found.** the uncertainties and limitations of the benefits analysis are discussed.

1.2.1 Site Descriptions

EPA modeled a variety of scenarios in order to obtain a broad range of the magnitudes of exposure reductions that might be seen nationally from a ban on lead wheel weights. EPA's goal was to obtain the largest range in areas modeled and include representative urban, rural, and suburban locations with differing housing vintages and background soil lead concentrations. Each case-study area is a generic composite urban, rural or suburban area, drawing on published and publically available data from real locations.

Variable characteristics in the scenario were:

- site type (i.e. urban, suburban or rural);
- traffic volume;
- housing vintage; and
- background soil lead concentration.

"Site type" categories were based on definitions from the U.S. Census Bureau (U.S. Census Bureau 2000b)[1]. EPA used the U.S. Census Bureau's definition of "urban area" to represent the urban locations

[1] "Urban Area" is synonymous the U.S. Census definition of an urban area: a land area that has a residential population of at least 50,000, an overall population density of at least 1,000 people per square mile and surrounding census blocks that have an overall density of at least 500 people per square mile. "Suburban area" is synonymous with the U.S. Census definition of an

(see Section 1.2.1.1 for definition); “urban cluster” to represent the suburban location (see Section 1.2.1.2); and “rural area” to represent the rural locations (see Section 1.2.1.3).

Traffic patterns varied based on the hypothetical location of interest; urban, suburban or rural. Using these three site types EPA captured a range of traffic volumes and fleet distributions. Additionally, average speeds in each site type were incorporated in the exposure assessment model.

Background soil lead concentrations and housing vintage were varied in order to obtain a comprehensive understanding of the impact of the ban in areas with high and low background lead levels. EPA’s case study scenarios include housing in the “pre-1940” and “post-1980” vintage categories in order to include homes that likely have the highest and lowest background lead levels. Homes built before 1940 will have the greatest amount of lead paint, whereas those built post-1980 will have the least amount of lead paint due to the evolution of lead content in paint over these time periods.

For background soil lead concentration, EPA had the option to choose from a continuous and infinite number of lead concentrations, given the large range of lead soil concentrations that exist in the United States (Aelion, Davis, et al., 2008 & Hynes, Maxfield et al., 2001). For simplicity, EPA categorized background soil lead levels into two groups, “high” and “low” background soil concentration. EPA defined “high” soil lead levels as being greater than 400 µg/g and “low” background soil lead levels as less than 400 µg/g. The threshold point between high and low background lead was determined to be 400 µg/g because it is the level at which EPA considers lead a hazard in a child’s play area (U.S. EPA 2001b). However, even with the delineation between high and low concentrations, EPA used a range of concentrations (12µg/g to 1,463 µg/g) in the analysis.

EPA considered varying the climate conditions for each location, but opted to hold the climate in each scenario constant. EPA assumed climate plays a negligible role in the exposure to lead wheel weights. Boston’s Logan Airport climate data was used for each scenario in the model because its precipitation patterns are representative of the national average for a majority of the year (National Weather Service, 2005).

The goal for the case-studies is to account for a large range of areas where benefits may be seen. Given the many possible combinations of the variables described above, the five hypothetical sites modeled include the following:[2]

(A) Urban area, high soil lead concentration, pre-1940 housing

(B) Urban area, high soil lead concentration, post-1980 housing

(C) Rural area, high soil lead concentration, pre-1940 housing

(D) Rural area, low soil lead concentration, post-1980 housing

(E) Suburban area, low soil lead concentration, post-1980 housing

The sections below describe each hypothetical location in more detail. The sites are also summarized in Table 2-1.

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urban cluster: the same as an urban area but with less than 50,000 people. “Rural” is anything that falls outside of the definition of an urban area or urban cluster.

[2] Note that traffic patterns correlate with site type.

1.2.1.1 Urban Sites

EPA used the U.S. Census Bureau's definition of urban area, "a land area that has a residential population of at least 50,000, an overall population density of at least 1,000 people per square mile and surrounding census blocks that have an overall density of at least 500 people per square mile" for the urban sites (U.S. Census Bureau 2000b). Urban locations contain 68 percent of the U.S. population (U.S. Census Bureau 2000a).

An annual average daily traffic count of 33,800 vehicles was used for the urban sites, and a speed of 35 miles per hour (mph). EPA used published data from a local transportation department to determine the traffic volume of an actual typical urban area (Boston Regional Metropolitan Planning Organization 2009), and then used Google Maps to confirm that traffic data was for an area near homes. This was done to ensure the reality of the hypothetical site; the modeled traffic data was representative of an area with a real potential for residential exposure to lead wheel weights. The fleet distribution for this road was modeled using the U.S. EPA MOVES model, which determined the mean vehicle weight is 2.8 metric tons (U.S. EPA, 2009c).

EPA used a background lead concentration of 1,463 µg/g in soil for the urban sites . This value is a concentration that is found in the published literature from home yard soil in an urban area and is the arithmetic mean lead concentration for the area analyzed (Hynes, Maxfield, et al., 2001). EPA recognizes that this is an exceptionally high lead concentration. However, the urban scenario are intended to serve as a high-end bound for background lead concentrations in order to estimate the full the range of potential benefits of the proposed rule.

Two urban sites were modeled; both used the same traffic, census and lead background data. The only difference between the sites is that one location used pre-1940 vintage homes and one used post-1980 vintage homes.

1.2.1.2 Suburban Site

One suburban area was modeled. The U.S. Census Bureau does not have an official definition for a suburban area and therefore EPA used the U.S. Census Bureau's definition of an "urban cluster" to represent the suburban location. An urban cluster (UC) is defined as the same as an urban area, but with less than 50,000 people (U.S. Census Bureau, 2000b). According to the U.S. Census, UCs contain about 11 percent of the U.S. population (U.S. Census Bureau, 2000a).

As with the urban site, EPA used published data from a state transportation department to determine the traffic volume in a typical suburban area and used Google Maps to confirm the traffic recorder data was collected on a street with homes nearby. The annual average daily traffic count for the suburban street was 3,100 vehicles and average speed was 30 mph (Massachusetts Department of Transportation, 2009). The MOVES model was used to estimate the mean vehicle weight in the suburban area: 4.2 metric tons (U.S. EPA, 2009c).

For the suburban area, EPA assumed a low background soil lead level of 37 µg/g, based on a yard soil lead concentration for a similar area, as reported by Schmitt, Trippler, et al. (1988). The housing vintage in this hypothetical location is post-1980.

1.2.1.3 Rural Sites

Two scenarios were based on hypothetical rural locations. The U.S. Census Bureau defines rural as anything that falls outside of the definition of an urban area or urban cluster (U.S. Census Bureau, 2000b). Twenty-one percent of the U.S. population lives in a rural area (U.S. Census Bureau, 2000a).

For the rural scenarios, EPA used an annual average daily traffic count of 755 vehicles per day and average speed of 25 mph. EPA obtained this traffic data from a state transportation department for an actual rural area, as collected by a traffic recorder placed in close proximity to residences (Montana Department of Transportation, 2009). The MOVES model was used to estimate the mean vehicle weight in the rural area: 5.6 metric tons (U.S. EPA, 2009c).

EPA modeled two rural hypothetical locations: one with low soil lead background concentrations and new homes (post-1980); and one with high soil lead background concentrations with old homes (pre-1940). The background lead soil concentration used for the area with the post-1980 homes is 12 µg/g, a lead soil concentration level that was recorded in a rural area (Aelion, Davis, et al., 2008). The background lead concentration used for the area with the pre-1940 homes is 656 µg/g, the maximum observed value by Schmitt, Trippler, et al. (1988) in non-urban areas. EPA realizes these are exceptionally low and high lead concentrations but are used in order to estimate the full the range of potential benefits of the proposed rule.

Table 2-1 presents a summary of each scenario considered in this analysis.

Table 2-1. Summary of Data Inputs for Hypothetical Locations that are used in the Benefits Assessment.[a]

Site Type	% of Total U.S. Population Living in Each Site Type	Background lead Concentration (µg/g)	Housing Vintage	Average Annual Daily Traffic Count (MPH)	Mean Vehicle Weight in Metric Tons
Urban	68%	1,463	Pre-1940	33,800 (35)	2.8
			Post-1980		
Suburban	11%	37	Post-1980	3,100 (30)	4.2
Rural	21%	656	Pre-1940	755 (25)	5.6
		12	Post-1980		

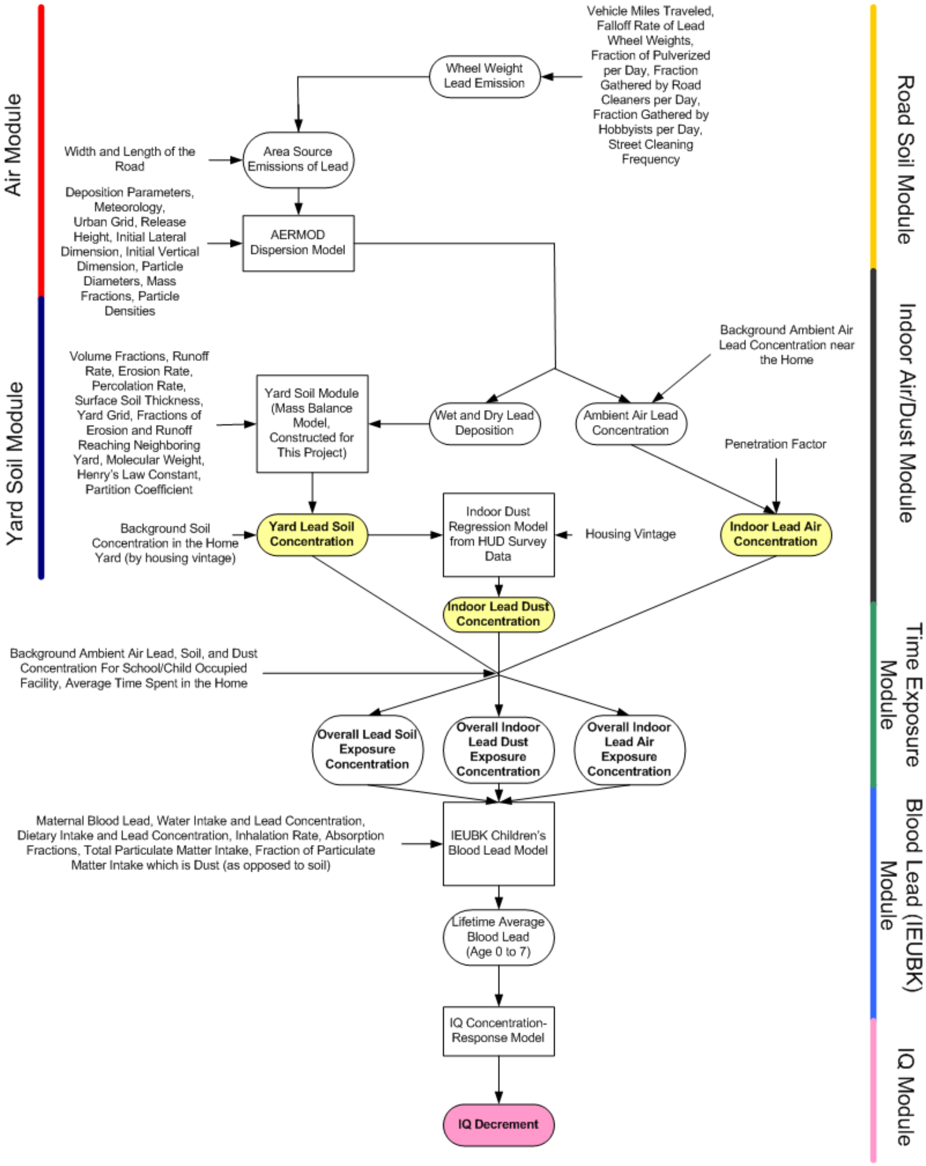
[a] All hypothetical locations use Boston Logan Airport climate data

Source: U.S. Census Bureau 2000a, Aelion, Davis, et al. 2009, Schmitt, Trippler, et al. 1988, Hynes, Maxfield, et al. 2001, ICF 2010, Montana Department of Transportation 2009, Massachusetts Department of Transportation 2009, Boston Regional Metropolitan Planning Organization 2009.

1.2.2 Exposure Modeling Approach, Inputs and Results

For each location, a series of exposure assessment modules were combined to model the exposure of lead from wheel weights due to inhalation and ingestion for a child aged zero to seven years. Figure 2-2 provides an overview of the modeling framework and the various inputs needed to complete the exposure assessment analysis.

Figure 2-2: Flowchart Showing the Modeling Framework for the Near-Roadway Residence and Exposure Scenario



Source: ICF 2010

The following sections provide a brief overview of each of the modules used to determine exposure to lead from wheel weights in each of the five scenarios. For a more detailed explanation of the approach used for this analysis see Appendix D.

1.2.2.1 Roadway Soil Module

The amount of lead emitted in the roadway dust is estimated using a mass-balance model. This amount of lead was estimated with information from Root (2000) which calculated lead wheel weight throw and pulverization rates. Additionally, a steady state assumption was made, meaning the reservoir of intact lead wheel weights by the roadway and the reservoir of pulverized lead in the roadway is not shrinking or growing. Thus, the amount of lead thrown from cars in the form of wheel weights equals the amount of lead pulverized each day, and this in turn equals the mass of lead emitted in roadway dust. In addition to this assumption, EPA also assumed there is a periodic removal of lead from the roadway during street cleaning events. These cleaning events lead to a periodic increase and decrease in the lead wheel weight reservoir on the road. In initial modeling efforts, street cleaning was not taken into account in the estimation of the lead emission rate. However, this resulted in a modeled concentration of lead wheel weights which exceeded the background lead concentration in the area. Given that the background concentration should include the contribution from lead wheel weights an assessment of the model resulted in the determination that street cleaning should be added to the model to ensure reasonable results. For additional information on how the street cleaning rates were determined see Appendix D.

The output of this module is the lead emission rate in roadway dust from the roadway per vehicle mile traveled. Table 2-2 provides the inputs needed for this module along with the values used and their source. For a detailed explanation on input value derivation, see Appendix D.

Table 2-2: Summary of Input Values for Roadway Soil Module

Variable Description	Units	Scenario A (Urban, high soil, early vintage)	Scenario B (Urban, high soil, late vintage)	Scenario C (Rural, high soil, early vintage)	Scenario D (Rural, low soil, late vintage)	Scenario E (Suburban, low soil, late vintage)	Source of Variable Value
Falloff rate of lead wheel weights	kg/VMT	1.56 E-6	1.56 E-6	1.56 E-6	1.56 E-6	1.56 E-6	Root (2000) with additional calculations
Fraction pulverized per day	`/day	0.0272	0.0272	0.0272	0.0272	0.0272	Root (2000)
Frequency of road cleaners	Cleanings per year	12	12	2	2	6	Professional Judgment
Fraction gathered by hobbyists per day	`/day	0	0	0	0	0	Professional Judgment

Source: ICF 2010.

1.2.2.2 Ambient Air Module

The ambient air concentration and yard lead deposition are estimated using the AERMOD dispersion model (U.S. EPA, 2009a). The case-study location is assumed to consist of a series of streets which intersect at regular intervals. Based on case-study locations for the urban, rural, and suburban scenarios described in Section 1.2.1, the block length, street width, and number of houses per block were used to create the emission grid (the roadways) and the receptor grids (individual yards). To account for different traffic patterns within a location, the grid contains both main arteries and residential streets, where each occur at specified regular intervals. The roadway dimensions, traffic patterns, and lead emissions are combined to estimate the area source of lead from the roadway. Meteorological conditions, land use information, and particulate attributes are also input into AERMOD for the dispersion calculation. The outputs of this module are the annual-average ambient air concentration, dry lead deposition, and wet lead deposition at the yard with the maximum exposure. For a detailed explanation on input value derivation, see Appendix D.

Table 2-3: Summary of Input Values for Ambient Air and AERMOD Module							
Variable Description	**Units**	**Scenario A (Urban, high soil, early vintage)**	**Scenario B (Urban, high soil, late vintage)**	**Scenario C (Rural, high soil, early vintage)**	**Scenario D (Rural, low soil, late vintage)**	**Scenario E (Suburban, low soil, late vintage)**	**Source of Variable Value**
Traffic volume on High Volume Streets	number of vehicles	33,800	33,800	755	755	3,100	Montana Department of Transportation 2009, Massachusetts Department of Transportation 2009, Boston Regional Planning 2009
Traffic volume on High Volume Streets	number of vehicles	8,450	8,450	189	189	775	Montana Department of Transportation 2009, Massachusetts Department of Transportation 2009, Boston Regional Planning 2009
Average length of city block in neighborhood of interest	m	150 x 60	150 x 60	115 x 115	115 x 115	200 x 105	Google Earth ®
Number of yards per city block	number	8 x 2	8 x 2	3 x 2	3 x 2	5 x 2	Google Earth ®
Average street width for each location	m	8	8	8	8	8	Google Earth ®
Wind speed, ambient temperature, mixing height, and turbulence parameters	various	From 12 months (Aug. 2009 through July 2010) of Logan Airport surface data and Chatham, MA upper-air data	NCDC Integrated Surface Hourly data, and NOAA/ESRL Radiosonde Database				Wind speed, ambient temperature, mixing height, and turbulence parameters
Months in each season	months	Winter: Dec., Jan., Feb Spring: Mar., Apr., May Summer: Jun., Jul., Aug. Autumn: Sep., Oct., Nov.	NCDC 1961-1990 and 1971-2000 Climate Normals, Hourly surface data from Boston Logan Airport				Months in each season

Land use Category	none	30% High Intensity Residential, 16% Commercial / Industrial/ Transport., 54% Urban/ Recreational Grasses	30% High Intensity Residential, 16% Commercial / Industrial/ Transport., 54% Urban/ Recreational Grasses	8% Low Intensity Residential, 13% Commercial / Industrial/ Transport., 80% Urban/ Recreational Grasses	8% Low Intensity Residential, 13% Commercial / Industrial/ Transport., 80% Urban/ Recreational Grasses	8% Low Intensity Residential, 11% Commercial / Industrial/ Transport., 81% Urban/ Recreational Grasses	Categories provided in AERMOD/AERMET guidance; percentages estimated from the spatial setups of the model scenarios
Release height	m	1.61	1.61	1.68	1.68	1.66	U.S. EPA (2010b) and information about fleet distribution from the MOVES model (U.S.EPA 2009c)
Initial vertical dimension	m	1.49	1.49	1.56	1.56	1.54	
Percentage of PM which is in the "fine" classification	Percent	0.626	0.626	0.626	0.626	0.626	Samara and Voutsa, 2005
Mass mean diameter of particulate	□m	0.85	0.85	0.85	0.85	0.85	Samara and Voutsa, 2005
Source: ICF 2010							

1.2.2.3 Yard Soil Module

The concentration of lead in the soil at the home of interest is calculated using a vertical mass balance model which assumes inputs into the upper soil layer from wet and dry deposition, as well as removal due to colloidal transfer out of the upper soil layer. Information on soil depth, density and porosity is needed along with an assumption on the residence time of lead and the background lead levels in soil. The inputs for these values are laid out in Table 2-4 below. For a detailed explanation on input value derivation, see Appendix D.

Table 2-4: Summary of Input Values for Yard Soil Module							
Variable Description	**Units**	**Scenario A (Urban, high soil, early vintage)**	**Scenario B (Urban, high soil, late vintage)**	**Scenario C (Rural, high soil, early vintage)**	**Scenario D (Rural, low soil, late vintage)**	**Scenario E (Suburban, low soil, late vintage)**	**Source of Variable Value**
Depth of surface soil compartment	M	0.01	0.01	0.01	0.01	0.01	Lower range of those provided in U.S. EPA (2009b)
Density of Soil Particle	kg/m^3	2,600	2,600	2,600	2,600	2,600	McKone et al. (2001)
Porosity of Soil	Fraction	0.2	0.2	0.2	0.2	0.2	McKone et al. (2001)
Residence time of lead in surface soil	Years	1,000	1,000	1,000	1,000	1,000	Conservative value based on literature search
Background home yard lead soil concentration	g/g	1,463	1,463	656	12	37	Literature citing yard soil measurements (Hynes, Maxfield et al., 2001; Schmitt, Trippler, et al. 1988; Aelion, Davis, et al., 2008 2008)
Depth of surface soil compartment	m	0.01	0.01	0.01	0.01	0.01	Lower range of those provided in U.S. EPA (2009b)
Density of Soil Particle	kg/m^3	2,600	2,600	2,600	2,600	2,600	McKone et al. (2001)
Porosity of Soil	fraction	0.2	0.2	0.2	0.2	0.2	McKone et al. (2001)
Source: ICF 2010							

1.2.2.4 Indoor Air/Dust Module

The indoor air/dust module estimates the indoor air lead concentration from the ambient concentration using a penetration factor developed by Thatcher and Layton (2005). It also estimates the indoor dust concentration using a regression model, the vintage of the home, and the calculated soil lead concentrations at the home. This regression model is used to account for the fact that indoor dust lead concentration is correlated with the outdoor soil (due to tracking of soil into the home) and the amount of lead in the indoor paint (which is itself correlated with the housing vintage). For a detailed explanation on input value derivation, see Appendix D.

Table 2-5: Summary of Input Values for Yard Soil Module							
Variable Description	**Units**	**Scenario A (Urban, high soil, early vintage)**	**Scenario B (Urban, high soil, late vintage)**	**Scenario C (Rural, high soil, early vintage)**	**Scenario D (Rural, low soil, late vintage)**	**Scenario E (Suburban, low soil, late vintage)**	**Source of Variable Value**
Background home ambient air concentration	□g/m3	0.025	0.025	0.010	0.010	0.014	Estimated from AQS monitoring network for 2008 (U.S. EPA, 2010a)
Penetration fraction of ambient air into home	fraction	1	1	1	1	1	Thatcher and Layton (1995)
Dust regression multiplier	none	59.0	38.3	59.0	38.3	38.3	Calculated from HUD Survey Data (U.S. EPA, 1995), incorporating the average paint concentration for each vintage)
Dust regression soil power	none	0.331	0.331	0.331	0.331	0.331	
Source: ICF 2010							

1.2.2.5 Time Exposure Module

The time exposure module accounts for the fact that the child is not in the home 24 hours per day. Background values are used to represent time outside the home, with the assumption made that the child is at a school or a child-occupied facility (such as a daycare center) during the time they are not at home. The relative fractions of time spent in the home and out of the home are used to combine the media concentrations into overall exposure concentrations for each year of the child's life, age zero to seven years.

For this assessment, it is assumed that if a child is outside the home, they are at a nearby school or daycare facility. Because these facilities are expected to be close to the child's home, the assumption was made that the exposure concentrations during the time outside the home are equal to the exposure concentrations in the home. (Another option was to assume *no* exposure was occurring at these facilities; this was considered unrealistic.) The assumption used, that the exposure is the same at home and at a nearby school or daycare facility is a recognized uncertainty and limitation of the model.

1.2.2.6 Resuspension

The modeling framework for the near-roadway residence includes resuspension of road dust into the air and the subsequent dispersion and deposition of this lead-containing dust into nearby yards. However, the actual model used in this analysis does not include the resuspension of contaminated yard soil into the air. In order to include this process, a full multi-media model which simultaneously models both air and soil processes would have to be used; these models tend to have less sophisticated dispersion algorithms than the air-only AERMOD model which was chosen for this analysis.

To determine the possible uncertainty associated with excluding yard soil resuspension, a literature search was conducted. The search focused on peer-reviewed journal articles which address resuspension of lead or other metals from soil to the air. The search was conducted using Google Scholar® and using search combinations of the following words: "lead", "metal", "resuspension", and "reentrainment". Appendix D lists the papers found in the literature search.

In general, the articles identified through this search suggest that resuspension of contaminated soil can be a large contributor to ambient air concentrations. Harris and Davidson (2009) employ a mass balance model to conclude that sources of lead due to the resuspension of contaminated soil/dust are a factor of ten higher than direct sources of lead in the South Coast Air Basin in California. They cite the main contributor of lead in the soil to be from historical deposition in the era of leaded gasoline, and the current sources due to resuspension include both yard soil and roadway soil. Sabin et al. (2006), however, found that much of the airborne lead in Los Angeles was due to resuspension from roadways, and concentrations of lead in air returned to near-background levels within 10 to 150 meters of the roadway. Hosiokangas et al. (2004) also found that roadways were a major contributor to airborne lead levels (27 percent) in Finland, and the windspeed tended to be the major determinant of how much lead was resuspended. Overall, these papers suggest that resuspension of contaminated soil/dust is a major contributor to airborne lead, but much of this resuspension occurs on roadways where car turbulence creates an effective mechanism for suspending the dust.

Thus, excluding yard resuspension will tend to under-predict the yard air lead concentrations; however, the dominant source to a yard next to a roadway is likely the resuspended roadway lead rather than lead resuspended from the yard itself. The exclusion of yard resuspension remains a recognized limitation of the modeling approach.

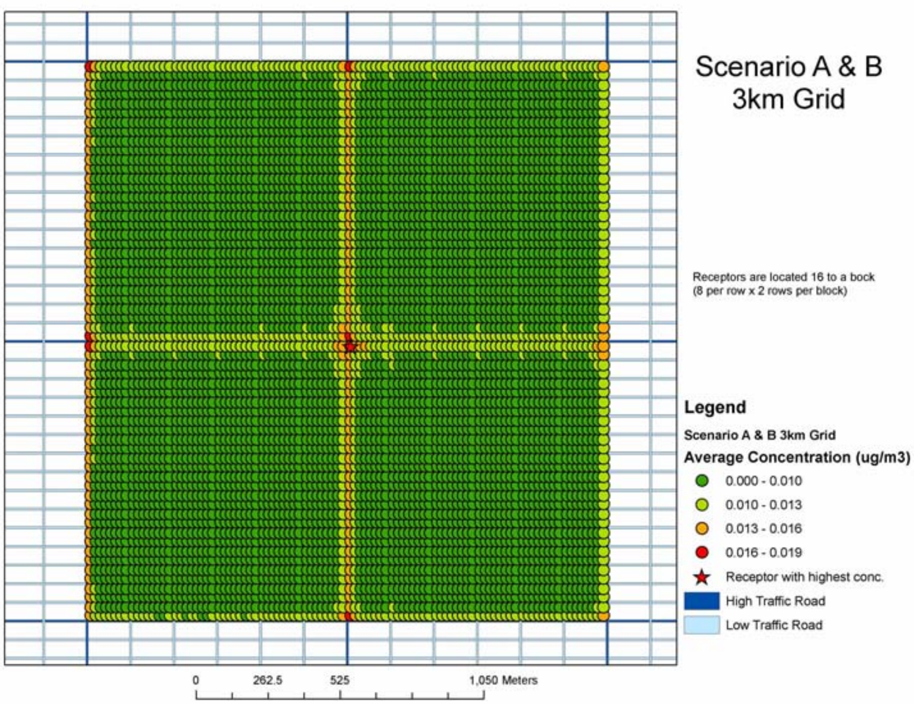
1.2.2.7 Media Concentration Results

Media lead concentrations for each scenario were determined by combining all of the scenario's exposure modules results. Scenarios A and B

The lead concentrations in the receptor yards relative to the high- and low- volume streets for the urban scenario three km grid are shown in Figure 2-3. When the model was first conducted, street cleaning was not included and the modeled lead concentration was 0.054 $\mu g/m^3$. This is above the background concentration and given that the background concentration should include the contribution from wheel weights, it was observed that the scenario was overly conservative. As discussed in Section 1.2.2.1, street cleaning was added to the model to ensure that more reasonable modeling results were achieved. For the remainder of the case-studies, street cleaning was included in the model.

In the revised model, the highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic and is indicated with a star. At this point, the concentration is 0.017 $\mu g/m^3$, and the total deposition (wet and dry) is 0.0011 g/m^2/year. The concentration of lead wheel weights is the same for both scenarios because the traffic data for both urban scenarios is the same. The modeled concentration can be compared with the background concentration of 0.025 $\mu g/m^3$ which includes the effect of lead wheel weights.

Figure 2-3: Total Yard Lead Concentrations for Both Urban Locations

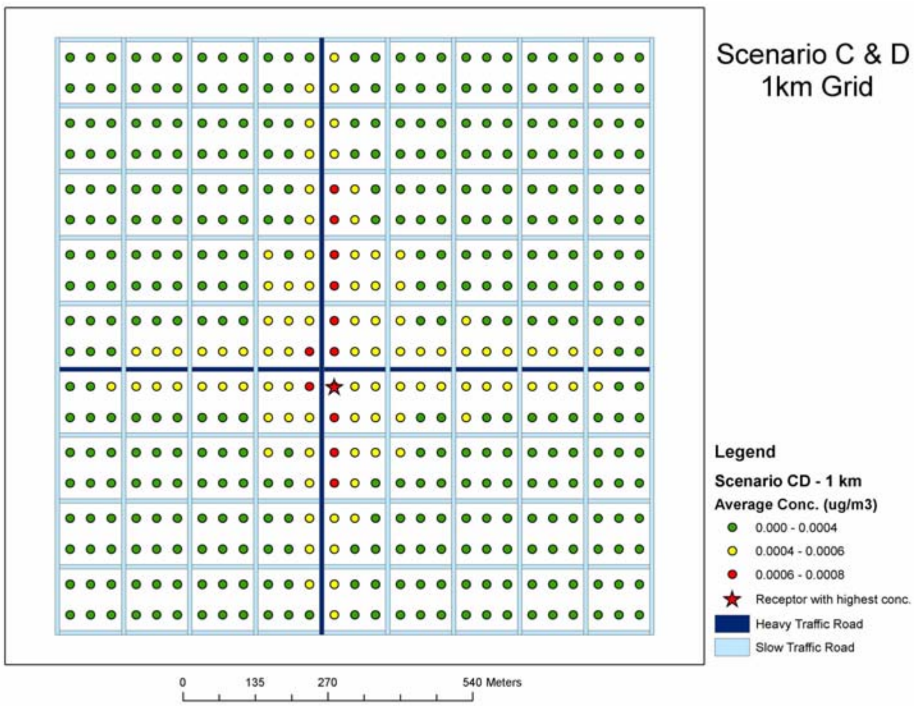


Source: ICF 2010

Scenarios C and D

The concentrations in the receptor yards relative to the high and low volume streets for the rural scenarios one km grid are shown in Figure 2-4. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 7.8E-4 □g/m^3, and the total deposition (wet and dry) is 5.3E-5 g/m^2/year. The concentration of lead wheel weights is the same for both scenarios because the traffic data for both urban scenarios is the same. The modeled concentration can be compared with the background concentration of 0.010 □g/m^3 which includes lead wheel weights.

Figure 2-4: Total Yard Lead Concentrations for Both Rural Scenarios

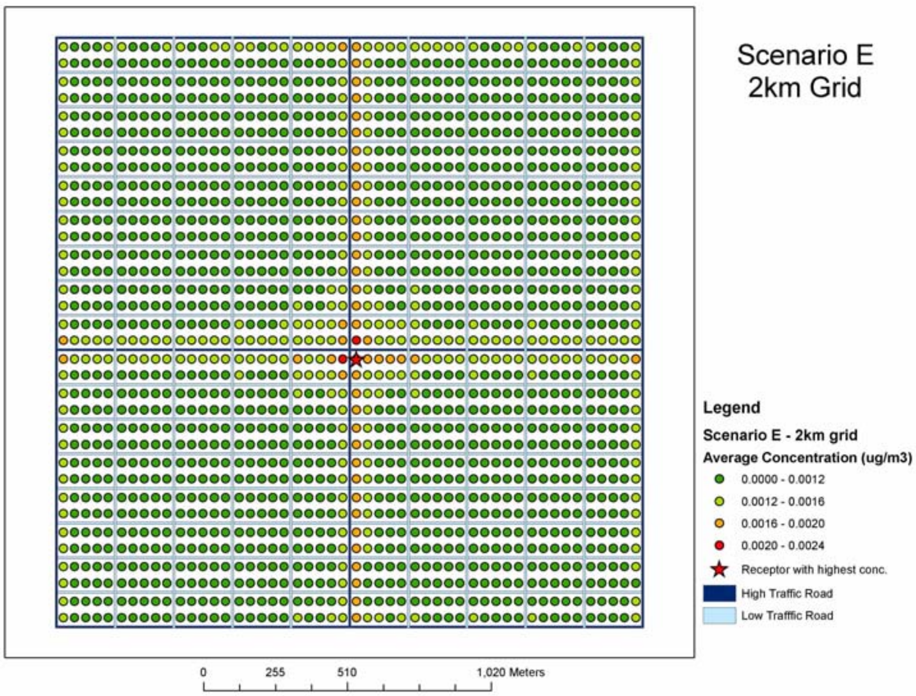


Source: ICF 2010

Scenario E

The concentrations in the receptor yards relative to the high and low volume streets for the suburban scenario two km grid are shown in Figure 2-5. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 2.1E-3 □g/m^3, and the total deposition (wet and dry) is 1.4E-4 g/m^2/year. The modeled concentration can be compared with the background concentration of 0.014 □g/m^3.which includes lead wheel weights.

Figure 2-5: Total Yard Lead Concentrations for the Suburban Scenario



Source: ICF 2010

1.2.2.8 Summary of Media Concentrations in the Modeled Scenarios

In each scenario, the modeled air concentrations were binned into intervals which span the range of modeled concentrations in the domain. The bins were selected so that each scenario had three or four bins and the bin boundaries were equally-spaced. Then, the percentage of yards in each concentration bin was calculated using all the modeled yards on the eastern side of the grid. Because the wind is predominantly from the western direction, the eastern side of the grid has a larger contribution from upwind wheel weight emission and thus has a higher level of concentration precision than the western side of the grid.

Next, the mean air concentration and deposition was calculated in each bin for each scenario. These concentrations were then used to calculate the soil and dust concentrations corresponding to these mean concentrations. The maximum air concentration and deposition in the domain were used to find the media concentrations at the maximally exposed home. Figure 2-5 shows these media concentrations calculated from the AERMOD modeling, the yard soil module, and the indoor dust module. The background estimates are presumed to include both the wheel weight and other lead source contributions. The wheel weight contribution in the table represents the portion of the total media concentration that is contributed by lead wheel weights. In the case of the dust concentration, this contribution is only approximate since the dust regression equation is nonlinear. The dust concentration was found using (1) the background soil concentration and (2) the background soil concentration minus the wheel weight contribution and then subtracting (2) from (1). In general, the wheel weight contributions are a small percentage of the total soil and dust concentrations, particularly in the high soil concentration and earlier housing vintage cases. The air concentration contribution is larger, varying from 8 percent in the rural case up to 70 percent in the urban case. The final media concentrations along with the proportion of yards in each bin are presented in Table 2-6.

Table 2-6: Media Concentrations in the Modeled Scenarios							
		Mean Concentrations					
Scenario	**Bin Number (Proportion of Yards Which Fall into corresponding Bin)**	**Background Air (µg/m^3)**	**Wheel Weight Contribution to Air (µg/m^3)**	**Background Soil (µg/g)**	**Wheel Weight Contribution to Soil (µg/g)**	**Background Dust (µg/g)**	**Approximate Wheel Weight Contribution to Dust (µg/g)**
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin 1 (85.9%)	0.0250	0.0083	1,463.0	25.0	658.5	3.7
	Bin 2 (11.6%)		0.0112		35.0		5.3
	Bin 3 (2.4%)		0.0142		44.8		6.7
	Bin 4 (0.1%)		0.0169		54.7		8.3
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin 1 (85.9%)	0.0250	0.0083	1,463.0	25.0	427.5	2.4
	Bin 2 (11.6%)		0.0112		35.0		3.4
	Bin 3 (2.4%)		0.0142		44.8		4.4
	Bin 4 (0.1%)		0.0169		54.7		5.4
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin 1 (76.7%)	0.0100	0.0003	656.0	0.9	504.9	0.2
	Bin 2 (19.6%)		0.0005		1.4		0.4
	Bin 3 (3.7%)		0.0007		2.1		0.5
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin 1 (76.7%)	0.0100	0.0003	12.0	0.9	87.2	2.2
	Bin 2 (19.6%)		0.0005		1.4		3.6
	Bin 3 (3.7%)		0.0007		2.1		5.4
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin 1 (79.3%)	0.0140	0.0010	37.0	2.7	126.6	3.1
	Bin 2 (15.9%)		0.0013		3.8		4.4
	Bin 3 (4.6%)		0.0018		5.4		6.4
	Bin 4 (0.2%)		0.0023		7.0		8.5
Source: EPA calculations, ICF 2010, Aelion, Davis, et al., 2008 , Schmitt, Trippler, et al. 1988, Hynes, Maxfield et al., 2001,							

1.2.2.9 Translating Media Concentrations to Blood Lead Levels

The media concentrations shown in Table 2-6 were input into the IEUBK blood lead model (U.S. EPA, 2010e) with the other inputs shown in Table 2-7.

Using this data the lifetime average blood lead value was calculated for the background case first. Then, the blood lead was calculated for each modeled scenario and bin by subtracting the wheel weight contribution to each media concentration from the background media concentration. In this way, the blood lead estimates represent situations where wheel weights are present and where wheel weights are not present, respectively. The results of this analysis are presented in Table 2-8.

Table 2-7: Blood Lead Model Input Values										
Group	**Parameter**	**Parameter Name**	**Parameter Value**							**Basis/Derivation**
			IEUBK Default Age Ranges (Years)							
			0.5 to 1	**1 to 2**	**2 to 3**	**3 to 4**	**4 to 5**	**5 to 6**	**6 to 7**	
Inhalation	Daily ventilation rate (cubic meters [m^3]/day)	Ventilation rate	5.4	8.0	9.5	10.9	10.9	10.9	12.4	U.S. EPA (2008b) Child-Specific Exposure Factors Handbook with interpolation for intermediate ages
Inhalation	Absolute inhalation absorption fraction (unitless)	Lung absorption	0.42							U.S. EPA (1989)
Inhalation	Indoor air Pb concentration	Indoor air Pb concentration (percentage of outdoor)	100%							These values are taken directly into account when developing the exposure concentrations
Inhalation	Time spent outdoors	Time spend outdoors (hours/day)	Not used							
Drinking Water Ingestion	Water consumption (L/day)	Water consumption (L/day)	0.36	0.271	0.317	0.349	0.380	0.397	0.414	U.S. EPA (2008b) Child-Specific Exposure Factors Handbook with interpolation for intermediate ages
Drinking Water Ingestion	Water Pb concentration (µg/L)	Lead concentration in drinking water (µg/L)	4.61							GM of values reported in studies of United States and Canadian populations (residential water) as cited in U.S. EPA (2006a), section 3.3 Table 3-10), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)
Drinking Water Ingestion	Absolute absorption (unitless)	Total percent accessible (IEUBK)	50 % (Single value used across all age ranges)							Assumed similar to dietary absorption (see "Total percent accessible" under Diet below), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)
Diet	Dietary Pb intake (µg/day)	Dietary Pb intake (µg/day)	3.16	2.6	2.87	2.74	2.61	2.74	2.99	Estimates based on the following: (1) Pb food residue data from U.S. Food and Drug Administration (U.S. FDA) Total Diet Study (USFDA, 2001), and (2) food consumption data from NHANES III (CDC, 1997), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)
Diet	Absolute absorption (unitless)	Total percent accessible	50%							Alexander et al. (1974) and Ziegler et al. (1978) as cited in U.S. EPA (2006, section 4.2.1), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)

Outdoor Soil/Dust and Indoor Dust Ingestion	Outdoor soil/dust and indoor dust weighting factor (unitless)	Outdoor soil/dust and indoor dust ingestion weighting factor (percent outdoor soil/dust)	45%							This is the percent of total ingestion that is outdoor soil/dust. Value reflects best judgment and consideration (results published by van Wijnen et al. (1990), as cited in (U.S. EPA, 1989), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)
	Total indoor dust + outdoor soil/dust ingestion (mg/day)	Amount of outdoor soil/dust and indoor dust ingested daily (mg)	60	110	110	110	110	110	110	U.S. EPA (2008b) Child-Specific Exposure Factors Handbook, excluding cases of soil-pica and geophagy
	Absolute gastrointestinal absorption (outdoor soil/dust and indoor dust) (unitless)	Total percent accessible (IEUBK)	0.30 for both outdoor soil/dust and indoor dust							Reflects evidence that Pb in indoor dust and outdoor soil/dust is as accessible as dietary Pb and that indoor dust and outdoor soil/dust ingestion may occur away from mealtimes (U.S. EPA 1989), as in the Lead NAAQS (U.S. EPA, 2007) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2008c)
Other	Maternal PbB (µg/dL)	Maternal PbB concentration at childbirth, µg/dL	0.847							NHANES 2007-2008, national weighted GM of all women aged 18-45 (CDC, 2009)
Source: ICF, 2010										

Table 2-8: Impact of Lead Wheel Weights on Blood Lead Concentration By Case-Study Location				
Case- Study Location	**Bin #**	**Baseline Lifetime Average Blood Lead Level (µg/dL)**	**Blood Lead Level with No Lead Wheel Weights (µg/dL)**	**Difference In Blood Lead (µg/dL)**
Location A: Urban area, high soil lead concentration, pre-1940 housing	1	9.79	9.676	0.114
	2		9.633	0.157
	3		9.591	0.199
	4		9.548	0.242
Location B: Urban area, high soil lead concentration, post-1980 housing	1	8.856	8.747	0.109
	2		8.696	0.160
	3		8.659	0.197
	4		8.621	0.235
Location C: Rural area, high soil lead concentration, pre-1940 housing	1	6.282	6.279	0.004
	2		6.277	0.006
	3		6.272	0.010
Location D: Rural area, low soil lead concentration, post-1980 housing	1	1.414	1.397	0.017
	2		1.387	0.027
	3		1.373	0.041
Location E: Suburban area, low soil lead concentration, post-1980 housing	1	1.756	1.726	0.030
	2		1.713	0.043
	3		1.694	0.062
	4		1.675	0.082
Source: ICF, 2010				

Because the background lead concentrations were determined at a case-study level, all individuals in a specific location have the same baseline blood lead level. However, the contribution of lead from wheel weights was modeled on a yard-by-yard basis which results in differences in blood lead levels with no wheel weights between the bins. Table 2-9 displays the mean concentration of lead in the soil in each bin and the mean lead deposition in each bin.

Table 2-9: Mean Concentration of Lead and Deposition from Lead Wheel Weights in Yards					
Case- Study Location	**Bin Number**	**Bin Mean Concentration ($\mu g/m^3$)**	**Bin Mean Deposition ($g/m^2/yr$)**	**Number of Yards with this Exposure**	**Percentage of Yards with this Exposure**
Location A: Urban area, high soil lead concentration, pre-1940 housing & Location B: Urban area, high soil lead concentration, post-1980 housing	1	0.00830	0.00052	2543	85.9%
	2	0.01122	0.00073	343	11.6%
	3	0.01418	0.00093	70	2.4%
	4	0.01693	0.00114	4	0.1%
Location C: Rural area, high soil lead concentration, pre-1940 housing & Location D: Rural area, low soil lead concentration, post-1980 housing	1	0.00030	0.000018	207	76.7%
	2	0.00047	0.000030	53	19.6%
	3	0.00066	0.000044	10	3.7%
Location E: Suburban area, low soil lead concentration, post-1980 housing	1	0.00098	0.000055	674	79.3%
	2	0.00132	0.000078	135	15.9%
	3	0.00178	0.000112	39	4.6%
	4	0.00227	0.000145	2	0.2%
Source: ICF 2010					

1.2.3 Quantified Blood Lead Level Impact on IQ

Various models are available that estimate the correlation between blood lead levels and IQ. Below is a summary of several studies which examine the quantitative relationship between blood lead and IQ level along with an explanation of how the model used in this analysis was chosen. Additionally, the results of translating blood lead levels to IQ with this model are presented.

1.2.3.1 Review of Literature on the Relationship Between Blood Lead Concentration and IQ

Surkan et al. (2007) assessed the relationship between blood lead levels less than 10 ug/dL in children aged six to ten years and cognitive abilities, including IQ. Children enrolled in the New England Children's Amalgam Trial from Boston, Massachusetts and Farmington, Maine were included in the study. After adjusting for covariates, including age, race, socioeconomic status, and primary caregiver IQ, children with 5 to 10 ug/dL had a 5 point lower IQ score when compared to children with blood lead levels of 1 to 2 ug/dL. Furthermore, Surkan et al. divided the Wechsler Achievement Test scores into reading and math scores. In adjusted analyses, children with blood lead levels of 5 to 10 ug/dL scored 7.8 and 6.9 points lower on reading and math composite scores, respectively, when compared to children with blood lead levels of 1to 2 ug/dL.

Schwartz (1992) conducted a meta-analysis assessing the strength of the association between blood lead and children's full-scale IQ in school age children by evaluating the results of eight studies in total and including both longitudinal and cross-sectional study types. Emphasis was given to the size of the effect, "since that allows comparisons that are informative about potential confounding and effect modifiers" (Schwartz, 1992). The author found that an increase in blood lead from 10 to 20 ug/dL was associated with a decrease of 2.6 IQ points.

Lanphear et al. (2005) examined the association of IQ test scores and blood lead concentration for children including those who had blood lead levels less than 10 µg/dL. The authors pooled the results from seven previous studies conducted from 1989 through 2003. Compared to other studies, Lanphear et al. (2005) evaluated the impact of elevated blood lead on IQ across a much larger population across several countries, and included the widest range of patterns of lead exposure and socioeconomic conditions. For the children evaluated, 18 percent had a maximum blood lead concentration less than 10 ug/dL and 8 percent had a maximum level less than 7 ug/dL. The geometric mean blood lead level peaked at 17.8 ug/dL and declined to 9.4 ug/dL by five to seven years of age (Lanphear et al., 2005).

Lanphear et al. (2005) used full-scale IQ scores, as indicated by a version of the Weschler Intelligence Scale for Children, as the primary outcome; the relationship between blood lead levels and IQ was assessed via a log-linear model. In developing the regression model, Lanphear et al. (2005) first tested whether the linear model applied in most of the cohort analyses provided a good fit for the wider range of blood lead levels presented in the pooled data. A restricted cubic spline function was fit to the data. Furthermore, the effects of available confounders individually and in combination were examined for modification of the IQ-blood lead relationship. These confounders included sex, birth order, birth weight, maternal education, maternal age, marital status, prenatal alcohol exposure and the Home Observation for Measurement of the Environment (HOME) Inventory score. The HOME inventory is "an index that reflects the quality and quantity of emotional and cognitive stimulation in the home environment" (Caldwell and Bradley, 1984 as cited in Lanphear et al., 2005).

An inverse relationship between blood lead concentration and IQ score was detected. An increase in concurrent blood lead from 2.4 to 30 ug/dL was found to be associated with a 6.9 point decrease in IQ (95 percent CI: 4.2-9.4). For an increase in blood lead from 2.4 to 10 ug/dL, a 3.9 point decrease (95 percent CI: 2.4-5.3) in IQ was observed, while an increase from 10 to 20 ug/dL resulted in a 1.9 point decrease in IQ (95 percent CI: 1.2-2.6) (Lanphear et al., 2005). Also, a 1.1 point decrease in IQ was associated with an increase in blood lead from 20 to 30 ug/dL (95 percent CI: 0.7-1.5).

While the Sukan et al. model is consistent with the Lanphear et al. model, it is slightly less comprehensive and only includes one geographic region; therefore, the Lanphear et al. model is preferred. Additionally, Schwartz's 1992 review is comprehensive and informative, however the Lanphear et al. study is more recent and is also widely accepted and previously used by EPA (US EPA 2008c). Due to these reasons, the Lanphear article was chosen as the basis for the model to translate blood lead levels to IQ in this analysis.

1.2.3.2 Modeled Relationship between Blood Lead and IQ

Lanphear et al. (2005) derived multiple regression equations that are used in this analysis to determine the relationship between blood lead and IQ. The models each have cutpoints where the slope of the concentration response curves change. The equations used for this analysis are presented below:

PbB < 1 IQ change = 0

PbB = 1 to 10 IQ change = PbB * -0.88

PbB > 10 IQ change = -8.8 + (PbB - 10) * -0.10

where: PbB = Lifetime average of the blood lead level

The results of entering the blood lead values into their respective regression equations to derive IQ decrements with and without lead wheel weights are displayed in Table 2-10.

Table 2-10: Life Average Blood Lead Levels and Subsequent IQ Changes at Baseline and Without Lead Wheel Weights, by Location.						
Case- Study Location	**Bin #**	**Baseline Lifetime Average Blood Lead Level (µg/dL)**	**Baseline IQ Change**	**Blood Lead Level with No Lead Wheel Weights (µg/dL)**	**IQ Change with No Lead Wheel Weights**	**Difference In IQ Change**
Location A: Urban area, high soil lead concentration, pre-1940 housing	1	9.79	-8.615	9.676	-8.515	0.100
	2			9.633	-8.477	0.138
	3			9.591	-8.440	0.175
	4			9.548	-8.403	0.213
Location B: Urban area, high soil lead concentration, post-1980 housing	1	8.856	-7.794	8.747	-7.698	0.096
	2			8.696	-7.653	0.141
	3			8.659	-7.620	0.174
	4			8.621	-7.587	0.207
Location C: Rural area, high soil lead concentration, pre-1940 housing	*1*	6.282	-5.528	6.279	-5.525	0.003
	2			6.277	-5.523	0.005
	3			6.272	-5.519	0.009
Location D: Rural area, low soil lead concentration, post-1980 housing	*1*	1.414	-1.244	1.397	-1.229	0.015
	2			1.387	-1.220	0.024
	3			1.373	-1.208	0.036
Location E: Suburban area, low soil lead concentration, post-1980 housing	*1*	1.756	-1.545	1.726	-1.519	0.027
	2			1.713	-1.508	0.038
	3			1.694	-1.491	0.055
	4			1.675	-1.474	0.071
Source: ICF 2010						

1.2.4 Time Period for Eliminating Lead Wheel Weight Use

The length of time it would take for lead wheel weights to no longer contribute lead to the environment was estimated using the United States Geologic Survey's report (2006) *Stock and Flows of Lead-Based Wheel Weights in the United States*, the Rubber Manufacturer's Association (RMA) (2006) *Tire Service Life: Study of Scrap Tires*, along with the Root study (2000). EPA recognizes that the RMA's (2006) presentation is not a peer-reviewed primary source, and therefore the Agency contacted the National Highway Transportation Safety Administration (NHSTA) to confirm the validity of the data. NHSTA confirmed this source as the most comprehensive data to date on the age distribution of tires.

The projected tonnage of lead from wheel weights in use in 2012 is calculated to be 45,000 tons, based on the number of registered vehicles in the United States and an approach similar to that used in the USGS publication *Stocks and Flows of Lead-Based Wheel Weights in the United States* (USGS, 2006). The method for deriving this value is outlined in Section **Error! Reference source not found.** of this analysis.

Additionally, USGS calculated that 8,000 tons of lead in wheel weights are in inventory that is replaced on an as-needed basis. It was assumed that all 8,000 tons of inventory will be added to the fleet of vehicles, through new cars or tires added to the road, after the first year of the ban.

EPA assumed there were three main pathways for removal of lead wheel weights from use on vehicles: (1) scrapping of old vehicles, (2) scrapping of old tires due to tire replacement and (3) accidental throw from the vehicle.

USGS estimated that 3,000 tons of lead wheel weights are removed per year through the scrapping of old vehicles, based on the number of automobiles, light trucks and SUVs scrapped in 2003. Data on scrapping of commercial trucks was not available, but according to USGS, the amount of lead in wheel weights is inconsequential and amounted to less than 500 tons. EPA assumed that the rate at which the lead wheel weights are removed through scrapping remains the same for each year, i.e. 0.067 (3,000/45,000 = 0.067) of lead wheel weights are recovered through scrapping every year, and is independent of the age of the tire. In other words, the rate of survival for lead wheel weights in regards to vehicle scrapping is 1-0.067 or 0.933.

As for lead wheel weights that fall off vehicles, it was calculated that 1.56×10^{-6} kg of lead wheel weights fall off per vehicle mile traveled previously in this analysis (see Appendix C). Combining this estimate with an estimate from the U.S. Department of Transportation (2010) that there were approximately 3 trillion vehicle miles traveled in 2008, EPA estimated that approximately 4,639 tons of wheel weights are thrown from cars each year. The information on the amount of lead wheel weights thrown from vehicles is highly uncertain given the lack of published data on this topic. EPA assumed that the rate at which the lead wheel weights are removed by falling off vehicles remains the same each year, i.e. 0.103 (4,639/45,000 = 0.103) of lead wheel weights fall off vehicles every year, in other words the rate at which lead wheel weights do not fall off tires is 1-0.103 or 0.897.

Lastly, for the tire survival rates, RMA (2006) surveyed over 14,000 scrapped tires to determine the age of the tire when it is scrapped. Their study displayed that tire survival rates are dependant on tire age. Table 2-11 shows the survival rates of tires based on tire age. EPA assumed these rates represent the tire survival rates for the entire tire stock on the road. Column B of the table displays the cumulative percent chance of survival, e.g.,. the chance that a tire will survive to be greater than two years old is 0.830. Column C represents the annual percent chance of survival, e.g., the chance that a two-year old tire will survive to be three years old is 0.858.

Table 2-11. Tire Survival Rates

Tire Age [a]	Cumulative Tire Survival Rate [b]	Annual Tire Survival Rate [c]
0	1.00	1.00
1	0.967	0.967
2	0.830	0.858
3	0.610	0.735
4	0.402	0.659
5	0.260	0.647
6	0.164	0.631
7	0.106	0.646
8	0.070	0.660
9	0.049	0.700
10	0.034	0.694
11	0.026	0.765
12	0.018	0.692
13	0.013	0.722
14	0.008	0.615
15	0.005	0.625
16	0.000	0.000

Note: For calculation purposes, EPA carries the values to three significant figures but realizes the estimates are not accurate to this level

Source: The cumulative percent chance of survival was given by RMA (2006), EPA calculated the annual percent chance of survival by dividing the cumulative percent chance of survival for one year by the previous year's cumulative percent chance of survival..

Table 2-12 presents the amount of lead from wheel weights remaining on the road over time as modeled following a ban. Column A shows the age of tires installed before the proposed ban. Column B shows the quantity of lead wheel weights in the beginning of the period. Column C shows the survival rate for a tire (see Table 2-11). Column D shows the probability that an installed wheel weight will not fall in a given year (0.897) and Column E shows the probability that a car will remain on the road in a given year (0.933). Column F shows that an additional 8,000 tons of lead wheel weights are expected to be added to cars from existing inventories after the ban. Column G is calculated as the product of columns B, C, D, and E plus the value in column F. Column H is the value presented in column G as a percentage of 45,000, the initial amount of lead from wheel weights on the road. Given that the tail of the distribution of tire life is drawn out by the small amount of tires that survive beyond 10 years, EPA assumed that a cut-off of five percent lead in wheel weights remaining was appropriate for this analysis.

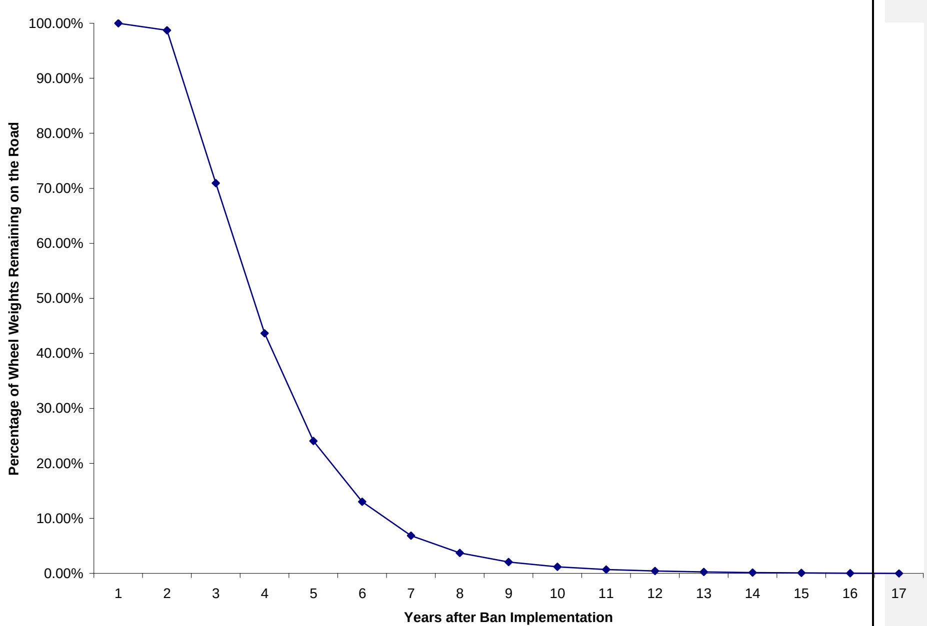
Table 2-12: Process for Determining the Percent of Lead Wheel Weights Remaining on the Road

Tire Age [a]	Amount of Lead from Wheel Weights on the Road at beginning of period (tons) [b]	Annual Tire Survival Rate [c]	Survival Rate of Lead Wheel Weights – *Fall Off* [d]	Survival Rate of Lead in Wheel Weights - *Scrapping* [e]	Lead Wheel Weights added to amount on the road due to stock [f]	Amount of Lead in Wheel Weights Remaining at end of period (tons) [g] = [b]x[c]x[d]x[e] + [f]	Percent of Lead Wheel Weights Remaining [h] = [g]/45,000
1	45,000	0.967	0.897	0.933	8,000	44,427	98.73%
2	44,427	0.858	0.897	0.933	0	31,922	70.94%
3	31,922	0.735	0.897	0.933	0	19,640	43.64%
4	19,640	0.659	0.897	0.933	0	10,835	24.08%
5	10,835	0.647	0.897	0.933	0	5,866	13.04%
6	5,866	0.631	0.897	0.933	0	3,098	6.88%
7	3,098	0.646	0.897	0.933	0	1,676	3.72%
8	1,676	0.660	0.897	0.933	0	927	2.06%
9	927	0.700	0.897	0.933	0	543	1.21%
10	543	0.694	0.897	0.933	0	315	0.70%
11	315	0.765	0.897	0.933	0	202	0.45%
12	202	0.692	0.897	0.933	0	117	0.26%
13	117	0.722	0.897	0.933	0	71	0.16%
14	71	0.615	0.897	0.933	0	36	0.08%
15	36	0.625	0.897	0.933	0	19	0.04%
16	19	0	0.897	0.933	0	0	0.00%

Source: USGS 2006, RMA 2006, Root 2000, EPA calculations.

A graph depicting the depletion of lead in wheel weights after a ban is implemented is displayed in Figure 2-6. The percentage of lead wheel weights remaining is based on the assumption that 45,000 tons of lead in wheel weights is the starting amount of lead in wheel weights when the ban is implemented.[3] After the first year of the ban there is a slight increase to over 100 percent of lead in wheel weights and this is due to the existing inventory being added into the market, increasing the amount of lead in wheel weights to greater than 45,000 tons. It can be seen by this graph that very little lead in wheel weights remain seven years after the implementation of a ban. The rule proposes to implement the ban in 2012 (year zero); therefore the benefits would begin to be seen in 2019.

Figure 2-6. Percent of Lead Wheel Weights Remaining Over Time



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3 EPA calculated this weight using the methodology described in Sections **Error! Reference source not found.** and **Error! Reference source not found.**. The tonnage of lead was derived by multiplying the estimated tonnage of weights in use by 95% to account for the 5% antimony in the alloy used for most lead wheel weights.

Appendix A - Calculation of Total Weight in 2003

As discussed in Section **Error! Reference source not found.**, the U.S. Geological Survey (USGS) estimated the total tonnage of lead wheel weights in use in 2003, as well as the tonnage of lead wheel weights manufactured in that year. The main cost analysis of this economic analysis uses a ratio of these tonnages to develop an estimate of lead-free weights manufactured as a result of a ban on lead wheel weights. EPA used the USGS estimate of manufactured weights, but calculated a new estimate of weight on the road using an approach similar to that used by USGS. This appendix provides the details of that calculation.

USGS estimated that 20,000 metric tons of lead weights were manufactured in 2003 (USGS, 2006). Although that report also included an estimated tonnage of lead wheel weights in use, EPA calculated this figure using a different assumption about the types of vehicles included in the registration data published by the Bureau of Transportation Statistics (BTS). The USGS analysis considered vehicles listed in the category "other 2-axle 4-tire vehicles" to be larger commercial vehicles having an average of seven ounces of balancing weight on each of the two front wheels. However, BTS defines this category as "[including] vans, pickup trucks, and sport utility vehicles." Other data from the Federal Highway Administration, which published the statistics used by BTS, also support the conclusion that this category is mainly composed of these types of light vehicles. Light trucks are balanced more similarly to passenger cars, and thus EPA applied the assumptions used for passenger cars to this category of vehicles.

The total tonnage of lead wheel weights on the road in 2003 was calculated by multiplying the number of registered vehicles in each category of vehicles by an assumed average amount of weight per wheel. USGS (2006) also assumed that 20 percent of passenger cars and light trucks were not using lead wheel weights in 2003, either because they used lead-free weights or no weights at all. Table A-1presents the resulting estimate of lead weight on the road in 2003.

Table A-1: Estimated Lead Wheel Weight on Registered Vehicles by Vehicle Type (2003)

Vehicle Type	Registered Vehicles 2003	% with Lead Weights	Weight per Wheel (ounces)	Wheels with Weights per Vehicle[a]	Total Weight (ounces)
Passenger car	135,669,897	80%	2.5	4	1,085,359,180
Other 2-axle 4-tire vehicle	87,186,663	80%	2.5	4	697,493,300
Truck, single-unit 2-axle 6-tire or more	5,848,523	100%	7.0	2	81,879,322
Truck, combination	1,908,365	100%	7.0	2	26,717,110
Bus	776,550	100%	7.0	2	10,871,700
Total	**231,389,998**				**1,902,320,612**

a. For large commercial vehicles, USGS (2006) noted the only significant weight to be on the two front wheels.
Source: USGS, 2006; U.S. BTS, 2010; EPA calculations

The weight shown in Table A-1was converted to 54,000 metric tons using a conversion factor of approximately 2.841 x 10^{-5} metric tons per ounce. EPA divided the 20,000 tons of lead weights manufactured in 2003 by the 54,000 tons of weight on the road. The resulting ratio of manufactured lead wheel weight to that in use is approximately 37 percent.

It should be noted that 54,000 metric tons represents the total weight of the whole wheel weight unit, which generally consists of 95 percent lead and 5 percent antimony. Multiplying total weight in use by 95 percent results in an estimate of lead in use in wheel weights in 2003, or approximately 51,000 metric tons.

Appendix B - Sensitivity Analysis: Disappearing Price Differential

This appendix addresses uncertainties inherent in the assumptions used for the cost analysis by examining the sensitivity of total costs to changes in those assumptions. The analysis presented in this section uses the alternative assumptions that the price differential between lead and lead-free wheel weights disappears after three and five years.

As explained in Section **Error! Reference source not found.**, EPA believes that the price differential between lead and lead-free wheel weights approximates the cost to manufacturers of complying with a ban. The main cost analysis assumes that this price differential will remain constant for the entire 20-year analysis period. However, EPA also believes that part of the current price differential for lead-free wheel weights reflects their current status as a lower-volume specialty item. The main cost analysis does not account for changes associated with higher production volumes, such as a change in the status of lead-free wheel weights to a non-specialty item, diminishing unit costs resulting from economies of scale, or production efficiency gains. To address this concern, EPA estimated total costs assuming that the price differential disappears after three and five years. Table B-1 presents the annual costs for the first five years of the rule. Total 20-year costs and annualized costs are presented for these two alternative assumptions in Table B-2.

Table B-1: Total Annual Costs in the First Five Years of the Rule

Discount Rate	Analysis Year				
	2012	2013	2014	2015	2016
Low Estimate					
Undiscounted	$16,296,732	$16,600,901	$16,914,444	$17,237,684	$17,570,952
3%	$15,361,233	$15,192,176	$15,028,265	$14,869,377	$14,715,396
7%	$14,234,197	$13,551,280	$12,903,949	$12,290,230	$11,708,268
High Estimate					
Undiscounted	$33,507,227	$34,129,772	$34,771,292	$35,432,436	$36,113,873
3%	$31,583,775	$31,233,576	$30,893,843	$30,564,330	$30,244,800
7%	$29,266,510	$27,860,060	$26,526,852	$25,262,837	$24,064,199

Source: EPA calculations.

Table B-2: Total 20-Year and Annualized Costs, Disappearing Price Differential

Discount Rate	After 3 Years		After 5 Years	
	Total 20-Year Cost	Total Annualized Cost	Total 20-Year Cost	Total Annualized Cost
Low Estimate				
Undiscounted	$49,812,078	$2,490,604	$84,620,714	$4,231,036
3%	$45,581,674	$3,063,804	$75,166,447	$5,052,366
7%	$40,689,426	$3,840,794	$64,687,924	$6,106,082
High Estimate				
Undiscounted	$102,408,291	$5,120,415	$173,954,600	$8,697,730
3%	$93,711,194	$6,298,864	$154,520,324	$10,386,193
7%	$83,653,422	$7,896,291	$132,980,458	$12,552,414

Source: EPA calculations.

Appendix C - Social Cost of Imported Lead-free Wheel Weights

This appendix presents an analysis of the costs associated with a ban on the import of lead wheel weights. While a ban on imports will not impose a direct cost to domestic manufacturers, it will likely have social costs associated with higher prices charged to consumers.

There are two main avenues through which lead wheel weights could be imported: (1) directly, for use on newly manufactured vehicles or for aftermarket tire balancing; and (2) indirectly, as parts used on imported vehicles.

EPA believes it is likely that most, if not all, foreign vehicle manufacturers with operations in the United States already use lead-free wheel weights on all newly manufactured vehicles. Because wheel weights are an insignificant portion of the overall vehicle cost, EPA assumes that any foreign vehicle manufacturers using lead weights will not pass the cost of lead-free weights on to consumers. Foreign manufacturers, therefore, would incur all of the costs for lead-free weights on vehicles imported into the U.S.

The second avenue for imported weights – direct importation – may have social cost implications for American consumers. As explained in the main cost analysis, EPA believes that nearly all of the increase in cost for producing lead-free wheel weights can be passed on to the service shops performing tire rebalancing. Assuming that all costs are passed through to the consumer, the price differential between imported lead and lead-free wheel weights can be used to estimate the social cost of a ban on imports.

To estimate the cost of a ban on the import of lead wheel weights, EPA followed a similar methodology to that used for the main cost analysis (see Sections **Error! Reference source not found.** – **Error! Reference source not found.**), and estimated the number of lead weights that would be imported in the absence of a ban. The U.S. Geological Survey (USGS) estimated that 4,000 metric tons of lead weights were imported in 2003 (USGS, 2006). In the main analysis, EPA assumed that half this amount was used by aftermarket tire balancing services. EPA estimated that approximately 4 percent of the tonnage of lead weights on the road in 2003 (54,000 metric tons) was imported for aftermarket use in that year.

Based on the assumption that the amount of imported lead weights is proportional to the number of registered vehicles, EPA used data from the Bureau of Transportation Statistics to forecast the annual number of lead weights on the road for the 20-year analysis period. EPA applied the ratio of imported weight to total weight on the road to determine the annual number of imported lead weights. Total costs were calculated by multiplying the total number of imported weights by the price differential between lead and lead-free weights. A more detailed description of this data and methodology is included in Sections **Error! Reference source not found.** – **Error! Reference source not found.**.

Table C-1 presents the resulting total social costs as a result of a ban on the import of lead wheel weights. Finally, Table C-2 shows the total 20-year and annualized costs.

Table C-1: Total Annual Social Costs of Imported Lead-Free Wheel Weights, 2012-2031					
Discount Rate	**Analysis Year**				
	2012	**2016**	**2021**	**2026**	**2031**
Low Estimate					
Undiscounted	$2,019,987	$2,172,887	$2,392,189	$2,648,179	$2,947,657
3%	$1,904,031	$1,819,759	$1,728,168	$1,650,258	$1,584,511
7%	$1,764,335	$1,447,886	$1,136,512	$897,030	$711,898
High Estimate					
Undiscounted	$4,148,616	$4,461,337	$4,909,233	$5,431,277	$6,041,133
3%	$3,910,468	$3,736,299	$3,546,534	$3,384,592	$3,247,406
7%	$3,623,562	$2,972,777	$2,332,341	$1,839,761	$1,459,013
Source: EPA calculations.					

Table C-2: Total 20-Year and Annualized Costs, Imported Lead-Free Wheel Weights		
Discount Rate	**Total 20-Year Cost**	**Total Annualized Cost**
Low Estimate		
Undiscounted	$48,818,674	$2,440,934
3%	$34,575,884	$2,324,043
7%	$23,131,772	$2,183,476
High Estimate		
Undiscounted	$100,164,209	$5,008,210
3%	$70,949,192	$4,768,900
7%	$47,472,508	$4,481,069
Source: EPA calculations.		

Appendix D - Derivation of Modeling Inputs and Models Used

Commented [AET1]: NOTE FOR EPA: This appendix is taken from the ICF draft report and included in the EA for convenient reference. In the final EA document we will reference the latest ICF report and will make the formatting consistent with the full content is included in the EA.

This appendix provides further detail on the exposure model approach and inputs. The exposure modeling for this proposed rulemaking was performed by ICF International. The text of this appendix is taken, verbatim, from their 2010 report, *Lead Wheel Weights Near-Roadway Exposure Analysis*, which is available in the docket for this rulemaking.

D.1 Ambient Air and AERMOD Module

Traffic Volume and AERMOD Street Grid

For each scenario, the traffic volume and street grid were determined using general attributes of urban, suburban, and rural cities, as described below. Average daily traffic values are shown in Table 2-13.

Scenario A and B

Based on examination of traffic counts in a Northeast proxy city provided by the state department of transportation, the highest traffic count on a road near residences was selected (33,800 vehicles per day). The modeled grid contains a series of intersecting roads separated by a distance equal to the block length (see Section III.A.2). It is unlikely that all roads in the residential area will have the high traffic counts. Thus, the traffic counts on a residential road in the same proxy city were determined, and the ratio between the low traffic street and the high traffic street was approximately 0.25. In addition, the higher volume streets occurred approximately every kilometer with lower volume streets between them. Thus, the model domain consists of a series of intersecting high volume streets with 33,800 vehicles per day every kilometer in both the north/south and east/west directions with lower volume streets with 8,450 vehicles per day spaced in the intervening blocks.

A sensitivity test was done to determine how far the grid of source streets should extend to capture the full contribution of wheel weights at the home of highest air concentration. This home occurs at the intersection of two busy streets near the center of the domain. Initially, a grid size of 1 km was used such that only a single intersection of high volume streets was present in the domain. However, increasing the domain extent to 3km (such that there were three high volume streets in the east-west and north-south directions) increased the air concentration at the maximally exposed home by 15%. However, the inclusion of the additional sources also significantly increased runtime. Because the wind direction is predominantly from the west-northwest (see Section III.A.4), an additional high volume street was added in the western direction, bringing the total extent in the east-west direction to 4km. In this case, the maximally exposed home increased by another 4%. However, it is likely that the presence of buildings would hamper the transport of lead emitted from the roadway at this distance. Thus, the concentrations from the 3km run were used in the analysis, and the 4km run suggests that these estimates could be underpredicting the lead concentration by up to 4% or more. Given the uncertainty in the emission rates, this degree of uncertainty was deemed acceptable for this modeling effort.

Scenario C and D

Traffic counts in a western proxy rural community were used to determine the traffic counts for these scenarios. The only available data were for a relatively high volume street through the town (755 vehicles per day). No data were available for the lower volume residential roads in the rural community. Thus, the same ratio between low volume and high volume streets used in the urban and suburban scenarios (0.25) was used to estimate a volume of 189 vehicles per day on low volume streets.

The extent of the rural community was approximately 1km with only a single high volume intersection. Thus, the model domain extended 1km in the north/south and east/west directions with a single high volume intersection in the middle of the domain, with lower volume streets spaced in between at intervals equal to the block length. No sensitivity test was done to increase the grid size, since it was deemed unlikely a rural town would extend further than 1km.

Scenario E

Traffic counts in a northeastern proxy suburban community were used to determine the traffic counts for this scenarios. The traffic counts for the highest traffic volume street near residences (3,100 vehicles per day) was selected. In addition, the ratio between a lower volume residential street and this high volume street were determined to be approximately 0.25. Thus, the same ratio between low volume and high volume streets used in the urban and rural scenarios (0.25) was used to estimate a volume of 775 vehicles per day on low volume streets.

Inspection of the pattern of roads indicated that higher volume streets occurred every 1km in the suburban community. Thus, the domain consisted of a 2km square with an intersection of higher volume roads in the center and higher volume roads along the perimeter, with lower volume roads along the intervening blocks.

Table 2-13. Estimate of Average Daily Traffic Counts by Road Type for Each Scenario.

Scenario	High Traffic Volume Average Daily Traffic (vehicles/day)	Low Traffic Volume Average Daily Traffic (vehicles/day)
A and B	33,800	8,450
C and D	755	189
E	3,100	775

Average Length, Number of Yards, and Street Width in City Block

The block layout for each scenario was determined by looking at proxy cities for each case. For the urban scenario, a northeast urban area was selected as a proxy. Measurement tools in GoogleEarth® were used to determine that a typical block is rectangular with the dimensions 150 x 60 m and the streets are 8 m wide. Visual inspection in GoogleEarth® revealed that there are typically 8 x 2 yards per rectangular block.

For the rural scenario, a western rural community was used as the proxy city. Measurement tools in GoogleEarth® were used to determine that a typical block is square with the dimensions 115 x 115 m and the streets are 8 m wide. Visual inspection in GoogleEarth® revealed that there are typically 3 x 2 yards per square block.

Finally, for the suburban scenario, a northeast suburban area was used as the proxy city. Measurement tools in GoogleEarth® were used to determine that a typical block is rectangular with the dimensions 200 x 105 m and the streets are 8 m wide. Visual inspection in GoogleEarth® revealed that there are typically 5 x 2 yards per square block.

Land Use Category and Surface Characteristics

AERMOD (specifically, the meteorological preprocessor, AERMET) requires the land use distributions of the study sites in order to estimate monthly values of three important surface characteristics (surface roughness length, albedo, and Bowen ratio). Surface roughness length contributes to AERMOD's estimations of surface turbulence and boundary layer stability. Albedo affects the amount of solar radiation absorbed by the surface, which affects boundary layer height and stability. The Bowen ratio describes how much surface heat is lost to the boundary layer through conduction and convection versus through evaporation, which affects the height and stability of the convective boundary layer.

AERMOD's land-use preprocessor, AERSURFACE, was developed to read in National Land Cover Database (NLCD) land use data (version 1992), calculate the distribution of land use types surrounding the study site, and use look-up tables where the values of the three surface characteristics depend on land use, season, snowfall, and rainfall amount. These surface characteristic look-up tables are available in Appendix A of the AERSURFACE User's Guide (USEPA, 2008a). The remainder of this section describes estimating the land use distribution of each study site. The climate information needed to determine seasons, snowiness, and rainfall amounts are described in Section A.7.

This study models a grid of city blocks that each have the same land use characteristics within the same study site (within the urban scenario, for example), rather than real locations with heterogeneous land use. As such, certain land use aspects of AERSURFACE (e.g., setting a land use radius for the surface roughness length, setting unique land use sectors) are not needed. Instead, the distribution of land use types surrounding the study sites was manually estimated, and, after also determining the climate characteristics, the look-up tables from USEPA (2008a) were used to estimate the values of the three surface characteristics.

The land area covered by residential buildings was estimated by first estimating the ground footprint of the typical residential building at each study site in this study (urban, suburban, and rural). Residential buildings include apartment buildings and attached and detached single family homes. The 2005 Residential Energy Consumption Survey results from the U.S. Energy Information Administration (USEIA, 2005) were used to estimate these footprints. Table 2-14 shows the estimated national number of the various types of residence buildings, the estimated percentage of each of these buildings at each of the study sites, and the estimated national average footprint of these buildings. The final column in Table 2-14 shows the assumptions that were made to estimate these numbers for this study. Note that towns are not used in this study but are shown in the table for completeness.

Table 2-15 shows the estimated average residence building footprint at each of the study sites. All of the footprints are between 190 and 205 m^2 (2,000 and 2,200 ft^2). Cities have the largest average footprint (203 m^2) due to a higher percentage of apartment buildings relative to single family homes, while rural areas have the smallest average footprint (193 m^2) due to a very small percentage of apartment buildings.

Table 2-14. Estimated U.S. Residence Building Characteristics

	National Total Count	% of National Total in… Cities	Suburbs	Towns	Rural Areas	Avg Footprint (m^2)	Assumptions
Detached Single Family Homes, 1 Floor	53,300,000	33%	22%	17%	27%	209	Detached single family homes include mobile homes, split-level, and 'other'
Detached Single family Homes, 2 Floors	24,000,000					161	
Detached Single family Homes, 3+ Floors	1,700,000					130	All have only 3 floors
Attached Single Family Homes, 1 Floor	2,600,000	64%	20%	16%	N/A	209	
Attached Single family Homes, 2 Floors	4,000,000					161	
Attached Single Family Homes, 3+ Floors	800,000					130	All have only 3 floors
Apartment Buildings, 2-4 Units, 1-2 Floors	1,950,000	67%	12%	18%	4%	304	All have 4 units; building count split evenly between 1 and 2 floors
Apartment Buildings, 5+ Units, 1-2 Floors	820,000	66%	16%	16%	2%	612	All have 10 units; building count split evenly between 1 and 2 floors
Apartment Buildings, 5+ Units, 3-4 Floors	300,000					470	All have 20 units; building count split evenly between 3 and 4 floors
Apartment Buildings, 5+ Units, 5-10 Floors	32,000					532	All have 50 units; building count split evenly between 5 through 10 floors
Apartment Buildings, 5+ Units, 11-20 Floors	600					259	All have 100 units; building count split evenly between 11 through 20 floors

* Note that the building characteristics for towns are not used in this study, but they are shown here for completeness. To convert m^2 to ft^2, divide by about 0.093.

Table 2-15. Estimated Footprint of the Average Residence Building in each Location Type

	Cities	Suburbs	Towns	Rural Areas
Avg Residence Building Footprint (m^2)	203	196	198	193

*Note that towns are not used in this study, but they are shown here for completeness. To convert m^2 to ft^2, divide by about 0.093.

Assuming that urban residential buildings tend to be taller than rural and suburban residential buildings, residential buildings for the urban study site were linked to the land use type “High Intensity Residential”, which is defined as “Includes highly developed areas where people reside in high numbers. Examples include apartment complexes and row houses. Vegetation accounts for less than 20 percent of the cover. Constructed materials account for 80 to100 percent of the cover” (USGS, 2010). Residential buildings for the rural and suburban study sites were linked to the land use type “Low Intensity Residential”, which is defined as “Includes areas with a mixture of constructed materials and vegetation. Constructed materials account for 30-80 percent of the cover. Vegetation may account for 20 to 70 percent of the cover. These areas most commonly include single-family housing units. Population densities will be lower than in high intensity residential areas” (USGS, 2010). These land use designations are also shown in Table 2-16. Table 2-16 also shows that cumulative footprint of residence buildings per city block, which was calculated by multiplying the average residence building footprint (Table 2-15) by the number of yards per city block (**Error! Reference source not found.**). The cumulative footprint of residence buildings per city block ranges from about 1,160 m^2 at the rural study site to about 3,251 m^2 at the urban study site.

For each study site, the land area covered by yards was estimated by subtracting the land area covered by residential buildings per city block from the area of each city block. The area of each city block was calculated by multiplying together the length and width of the city block (**Error! Reference source not found.**). Yards were linked to the land-use type “Urban/Recreational Grasses”, which is defined as “Vegetation (primarily grasses) planted in developed settings for recreation, erosion control, or aesthetic purposes. Examples include parks, lawns, golf courses, airport grasses, and industrial site grasses” (USGS, 2010). This land use designation is shown in Table 2-16, which also shows that the cumulative yard area per city block ranges from about 5,749 m^2 at the urban study site to about 19,040 m^2 at the suburban study site.

For each study site, the land area covered by roads per city block was calculated by allocating to the block half the width of each road bordering the block. Roads were linked to the land use type “Commercial/Industrial/Transportation”, which is defined as “Includes infrastructure (e.g. roads, railroads, etc.) and all highly developed areas not classified as High Intensity Residential” (USGS, 2010). AERSURFACE requires the user to specify if the study site is at an airport, which, if answered yes, would lower the surface roughness lengths otherwise associated with the “Commercial /

Industrial / Transportation" land use type. For all three study sites, the site is specified as non-airport. This land use designation is shown in Table 2-16, which also shows that the cumulative road area per city block ranges from about 1,744 m^2 at the urban study site to about 2,504 m^2 at the suburban study site.

Table 2-16. The Land Use Characteristics of Each Study Site

	Urban Study Site	Rural Study Site	Suburban Study Site
Area of City Block, Including Half of Roads on Every Side (m^2)	10,744	15,129	23,504
Cumulative Area of Residence Buildings per City Block (m^2)	3,251	1,160	1,960
% of Area of City Block that is Comprised of Residence Buildings	30%	8%	8%
Land Use Type for Residence Buildings	High Intensity Residential	Low Intensity Residential	Low Intensity Residential
Cumulative Area of Yards per City Block (m^2)	5,749	12,065	19,040
% of Area of City Block that is Comprised of Yards	54%	80%	81%
Land Use Type for Yards	Urban/ Recreational Grasses	Urban/ Recreational Grasses	Urban/ Recreational Grasses
Cumulative Area of Roads per City Block, With Half of Roads Included on Every Side (m^2)	1,744	1,904	2,504
% of Area of City Block that is Comprised of Roads	16%	13%	11%
Land Use Type for Roads	Commercial/Industrial/Transportation (non-airport)	Commercial/Industrial/Transportation (non-airport)	Commercial/Industrial/Transportation (non-airport)

* The land use types correspond to those contained in the 1992 NLCD (USGS, 2010).

These land use distributions are combined with season and rainfall information to determine the monthly values of the three surface characteristics. The climate information needed to determine seasons and rainfall quantities is described in Section III.A.7.

Meteorology Parameters and Length of Each Season

Because meteorological conditions are not expected to strongly affect exposure levels, all three scenario locations use meteorological data from Boston Logan International

Airport. AERMET requires hourly surface data and twice-daily upper-air data. The hourly surface data for Boston Logan International Airport (Weather-Bureau-Army-Nave (WBAN) identifier 14739) were obtained from National Climatic Data Center (NCDC) and are in Integrated Surface Data Tape Data-3505 format (NCDC ISD, 2010). These surface hourly data were formatted as necessary for use in AERMET, and only the official end-of-hour observations were used. The wind direction was predominantly from the west-northwest, as shown in Figure 7. The closest upper-air station to Boston Logan International Airport is located in Chatham, MA (WBAN identifier 14684). The upper-air data for Chatham were obtained from the National Oceanic and Atmospheric Administration/Earth System Research Laboratory Radiosonde Database Access (NOAA ESRL, 2010). The upper-air data are in AERMET-friendly Forecast Systems Laboratory format, and only the official 00 Coordinated Universal Time (UTC) and 12 UTC observations at mandatory and significant atmospheric levels will be used. In order to model air concentrations and deposition using the most recent 12-month meteorological data, the surface and upper-air data were obtained for August 2009 through July 2010.

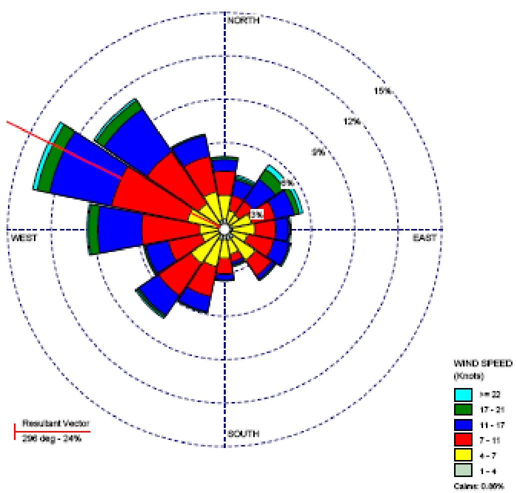


Figure 7. Wind Rose for Boston Logan Airport Meteorologic Station

As described in Section III.A.6, AERMET also requires three important surface characteristics – surface roughness length, albedo, and Bowen ratio. AERSURFACE uses land use data and user-specified climate information to estimate the monthly values of these three surface characteristics. Section III.A.6 describes the estimation of land use distribution for the study sites. In this section, the climate information is described.

The values of the surface characteristics for a given land use type can vary by season, so the user must define the seasons of the study sites. Because Boston Logan International Airport is being used as the meteorological proxy for this study, the climatology of the airport area was analyzed in order to define which month is part of which season.

First, winter must be defined as snowy or not snowy, where snowy is defined as experiencing continuous snow cover for at least one month per year. Defining winter as snowy tends to increase wintertime albedo, decrease Bowen ratio, and decrease surface roughness length for most land use types, compared to non-snowy winters. As described in USEPA (2009d), the shapefiles from the NCDC Climate Maps of the United States database (NCDC, 2005a) were used to analyze typical snow cover at any location in the lower 48 U.S. states. The shapefiles represent 1961-1990 Climate Normals and are contoured at specific intervals of values. Since AERSURFACE's definition of snowy is a month of continuous snow cover, the most analogous shapefile contour interval was the one defining at least 28.5 days of at least inch (25.4 mm) of ground snow depth. For simplicity, these 28.5 days were assumed to be continuous. The Boston Logan International Airport location met this definition of snowy.

Second, each month must be assigned to a season. The same procedures used in USEPA (2009d) to determine seasons for the lower 48 U.S. states were used in this study. As with defining continuous snow cover, the procedures for defining seasons relied on data from NCDC (2005a). The season assignments are described below. Based on these criteria, winter at the Boston Logan International Airport location was defined as December through February, spring was defined as March through May, summer was defined as June through August, and autumn was defined as September through November.

Finally, the AERSURFACE look-up tables require information as to whether the location was experiencing above average, below average, or average precipitation on a monthly basis. To determine the precipitation category, the AERSURFACE guidance recommends comparing the period of record of the meteorology data used in the modeling to the 30-year period of record for the same location and selecting above average if the modeling period is in the upper 30th percentile of the 30-year record, below average if in the lower 30th percentile, and average if otherwise. AERSURFACE applies this precipitation designation to the whole period of modeling. For the August 2009 through July 2010 period of modeling for this study, the 12-month total precipitation was 53.44 inches (135.7 cm) at the Boston Logan International Airport, which is 26% above the 1971-2000 Climate Normals annual precipitation amount of 42.53 inches (108 cm) (NCDC, 2005b).

However, individual months of the period of modeling range from 49% drier than normal to over 300% wetter than normal. Because this study will calculate monthly values of surface roughness length, albedo, and Bowen ratio, and because of these large monthly variances in precipitation, it is useful to categorize the precipitation amounts on a monthly basis. Monthly precipitation categories were used also used in the NO_2 NAAQS risk analysis (USEPA, 2008d), where AERSURFACE was run three times (once per precipitation setting), and the monthly values of the three surface characteristics using the three precipitation settings were merged according to monthly precipitation.

Monthly precipitation amounts from NWS (2005) were compared against the August 2009 through July 2010 monthly precipitation amounts. As shown in Table 2-17, two of the 2009-2010 months experienced precipitation amounts that were less than their respective 30th percentile 1971-2000 values. Three of the months experienced precipitation amounts that were greater than their respective 70th percentile 1971-2000 values. The other seven months experienced precipitation amounts that were within their respective 30th and 70th percentile values.

Table 2-17. Comparison of Monthly Precipitation to Average Conditions to Determine Precipitation Category

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Ann
August 2009-July 2010 Monthly Precipitation Amount (cm)	6.10	7.59	39.98	4.65	8.51	11.56	7.24	8.41	8.00	14.27	9.32	10.11	135.74
1971-2000 70th Percentile Monthly Precipitation Amount (cm)	12.30	8.98	10.89	11.16	9.13	7.75	10.15	11.08	11.66	10.87	13.03	12.88	115.61
1971-2000 30th Percentile Monthly Precipitation Amount (cm)	6.35	6.33	6.36	6.05	5.60	3.79	5.46	4.06	3.98	7.48	6.26	5.69	96.09
"Wetness" Category for 2009-2010 Data (used for AERSURFACE)	DRY	AVG	WET	DRY	AVG	WET	AVG	AVG	AVG	WET	AVG	AVG	WET
Season	Winter (snowy)	Winter (snowy)	Spring	Spring	Spring	Summer	Summer	Summer	Autumn	Autumn	Autumn	Winter (snowy)	--

The culmination of the land use and climate characteristics described in this section and Section III.A.6 is shown in Table 2-18. Table 2-18 shows the values of the three surface characteristics (albedo, Bowen ratio, and surface roughness length) for each month and for each scenario location type (urban, rural, and suburban). For each location type, these values were determined by averaging together the values of each surface characteristic for each land use type specific to the location. The averaging is weighted by the area of each land use type per city block. The surface characteristic value look-up tables are provided in Appendix A of the AERSURFACE User's Guide (USEPA, 2008a). The areas of each land use type per study site are shown in Table 2-16, and the season and "wetness" category assigned to each month are shown in Table 2-17.

Table 2-18. Model Values of Albedo, Bowen Ratio, and Surface Roughness Length for each of the Three Study Scenario Types

Month	Season	"Wetness" Category	Albedo			Bowen Ratio			Surface Roughness Length (m)		
			Urban	Rural	Suburban	Urban	Rural	Suburban	Urban	Rural	Suburban
Jan	Winter	Dry	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13
Feb	Winter	Avg	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13
Mar	Spring	Wet	0.48	0.15	0.15	0.57	0.33	0.32	0.44	0.15	0.14
Apr	Spring	Dry	0.48	0.15	0.15	1.93	1.33	1.30	0.44	0.15	0.14
May	Spring	Avg	0.48	0.15	0.15	0.86	0.49	0.47	0.44	0.15	0.14
Jun	Summer	Wet	0.16	0.15	0.15	0.63	0.41	0.40	0.44	0.16	0.15
Jul	Summer	Avg	0.16	0.15	0.15	0.96	0.65	0.63	0.44	0.16	0.15
Aug	Summer	Avg	0.16	0.15	0.15	0.96	0.65	0.63	0.44	0.16	0.15
Sep	Autumn	Avg	0.16	0.15	0.15	1.07	0.82	0.81	0.44	0.15	0.14
Oct	Autumn	Wet	0.16	0.15	0.15	0.68	0.49	0.48	0.44	0.15	0.14
Nov	Autumn	Avg	0.16	0.15	0.15	1.07	0.82	0.81	0.44	0.15	0.14
Dec	Winter	Avg	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13

* These values were derived from the tables in Appendix A of the AERSURFACE User's Guide (USEPA, 2008a), along with the "wetness" and season designations shown in Table 2-14 and the land use characteristics shown in Table 2-16.

Release Height and Dimensions

Road sources will be modeled in AERMOD as area sources. According to 40 CFR Part 5: "Revision to the Guideline on Air Quality Models: Adoption of a Preferred General Purpose (Flat and Complex Terrain) Dispersion Model and Other Revisions; Final Rule," re-entrained dust from roadway sources can be modeled as area, volume, or line sources (USEPA 2005b, page 68235). Area sources were selected to be consistent with the Office of Transportation and Air Quality's (OTAQ) "Development and Evaluation of an Air Quality Modeling Approach for Lead Emissions from Piston-Engine Aircraft Operating on Leaded Aviation Gasoline." OTAQ's lead aircraft assessment modeled three roadways adjacent to the airport using this methodology (USEPA 2010b, page 49).

AERMOD requires the following parameters to be assigned for each source: Emission Rate (Aermis), Release height (Relhgt), width of roadway (Xinit) and initial vertical dimension (Szinit) (USEPA 2004). Average release heights and initial vertical dimensions for light-duty and heavy duty vehicles are presented in "Transportation Conformity Guidance for Quantitative Hot-spot Analysis in $PM_{2.5}$ and PM_{10} Nonattainment and Maintenance Areas" (USEPA 2010d). Table 2-19 below lists default values by vehicle type. Site specific vehicle type distributions were obtained from MOVES (USEPA, 2009c) and a class-weighted average was applied to get site-specific release height and initial vertical dimension values for each scenario (see Table 2-20). This method is consistent with USEPA (2010d) recommendations.

Table 2-19. Default Release Height and Initial Vertical Dimension for AERMOD modeling

Vehicle Type	Release Height (Relhgt)	Initial Vertical Dimension (Szinit)
Light-duty	1.3 m	1.2 m
Heavy-duty	3.4 m	3.2 m

Table 2-20. Calculation of Release Height and Sigma Z for Scenarios A-E

Location	Light-duty vehicle distribution*	Heavy-duty vehicle distribution*	Release Height (m)	Sigma Z (m)
Suffolk, MA (Scenario A,B)	85.3%	14.7%	= (1.3×0.853) + (3.4×0.147) = **1.61 m**	= (1.2×0.853) + (3.2×0.147) = **1.49 m**
Boulder, MT (Scenario C,D)	81.8%	18.2%	= (1.3×0.818) + (3.4×0.182) = **1.68 m**	= (1.2×0.818) + (3.2×0.182) = **1.56 m**
Franklin, MA (Scenario E)	82.8%	17.2%	= (1.3×0.828) + (3.4×0.172) = **1.66 m**	= (1.2×0.828) + (3.2×0.172) = **1.54 m**

* Calculated from MOVES; Heavy Duty is the sum of vehicle population for Combination Long-haul Truck, Combination Short-haul Truck, Intercity Bus, Light Commercial Truck, Motor Home, School Bus, Single Unit Long-haul Truck, Single Unit Short-haul truck, and Transit Bus divided by the total population; Light-duty is the sum of vehicle population for Motorcycle, Passenger Car and Passenger Truck divided by the total vehicle population.

Percentage of Particulate Matter in the Fine Classification and Mean Mass Diameter

A requirement of AERMOD deposition Method 2 is the fraction of fine particulate matter (< 2.5 μm) in total particulate matter for the road-dust which will be modeled and the mass-median particle diameter (MMAD). Samara and Voutsa (2005) reported size distributions of roadside particulate matter and the MMAD near a roadway in Thessaloniki, Greece. The average mass-median particle diameter was 0.85 ± 0.71 μm. Samara and Voutsa (2005) reported average concentrations of roadway dust for the following size-bins:

Average concentration of PM by size (N=32), in μg/m^3:

< 0.8 μm: 54.2 ± 22.2

0.8 – 1.3 μm: 6.59 ± 6.79

1.3 – 2.7 μm: 5.68 ± 3.37

2.7 – 6.7 μm: 16.7 ± 9.34

> 6.7 µm: 23.0 ± 14.3

To calculate the fraction of fine particulate matter, the of average concentrations in size bins below 2.7 µm were summed and divided by the sum of concentrations in all bins. This results in a fraction of fine particulate matter of 0.626 for road dust.

Roadway Soil module

alloff rate of lead wheel weights

The fall off rate of lead wheel weights is derived from information presented in Root (2000). The study estimates wheel weight lead deposition along the 2.4 km six-lane divided "JTML" road in Albuquerque, New Mexico at 11.8 kg/km/year . The study notes that this estimate represents the deposition along the outer curb of both sides of the street. The study also observes that the median side deposition amounts to 25% of the curb side deposition at steady state. To include deposition along the median edge of both sides of the divided street, the curb side deposition estimate was multiplied by a factor of 1.25 in order to estimate deposition for the entire street. Accordingly, it was assumed that the lead wheel weight deposition was 1.25 x 11.8 = 14.75 kg/km/year, which is equivalent to 23.6 kg/mile/year along that street segment. To normalize the lead wheel weight deposition rate by the vehicle miles traveled, an average daily traffic flow of 41,500 vehicles/day was used, which is the traffic flow rate for the surveyed JTML street segment as cited in the Root study. The estimated normalized wheel weight lead deposition rate is therefore equal to 23.6/(41500 x 365) = 1.56 E-6 kg/VMT. This deposition rate was multiplied by the vehicle counts in Section III.A.1 to estimate the total mass per mile traveled. By making this calculation, it was assumed that all vehicles on the road are equally likely to eject wheel weights. In reality, only passenger vehicles will likely have wheel weights. However, as long as the mix of traffic along the JTML Root study road and the modeled road are similar, this assumption will not significantly bias the results.

Fraction of Wheel Weights Pulverized Each Day

The fraction of lead wheel weights pulverized per day is also obtained from the Root (2000) study, where it is estimated at 0.0272 or 2.72%. Although the Root study has numerous limitations, as noted in EPA (2005a), no superior study on the subject could be found despite an extensive literature search. Among other deficiencies, the study suffers from the significant limitation that it does not account for street cleaning and wheel weight collection by hobbyists, and does not examine any other potential fate and transport processes for the lead wheel weights. Root's estimate of the pulverization rate is in fact an estimate of the sum of all loss processes that wheel weights are subject to. It therefore represents an overestimate or an upper bound for the true pulverization rate. In the absence of better designed studies on the subject, a daily pulverization rate of 2.72% was used, while noting that it represents a conservative estimate of the true pulverization rate.

Street Cleaning Frequency

Street cleaning is a significant loss process impacting the stock of wheel weights on a road. Ignoring the effects of street cleaning would overestimate the risks from lead wheel weights. To determine the typical frequency of street cleaning, statistics of street cleaning from various cities were pulled from a compiled report (Schilling, 2005). The statistics show the frequency of street cleaning for a main artery, a central business district, and a residential area. Because the modeling domain includes the intersection of two busy streets in the urban, suburban, and rural scenarios and the highest concentration occurs at the crossroads, the central business district statistics were selected as the best descriptor of cleaning frequency. These frequencies are higher than in the purely residential area but reflect probable cleaning frequencies for high volume roads near residential areas.

For each city, the population, population density, and city type were determined from census information. The population corresponds to census information from 2006 while the population density (persons per square mile) corresponds to census information from 2000 (http://quickfacts.census.gov/qfd/). Using the population and population density, each city was mapped to a city type using the following census definitions:

1. **Urban Area (UA): 500 people per square mile with at least 50,000 people.**
2. **Urban Cluster (UC): 500 people per square mile with a population of at least 2,500 people, but fewer than 50,000 people.**
3. **Rural: anything outside of the definition of UC or UA**

If a city did not have available population density information, the population alone was used to map the city to a classification. Then, the UA designation was used to capture the urban areas of modeling scenarios A and B, the UC designation was used to capture the suburban areas in modeling scenario E, and the rural designation was used to capture the rural areas in modeling scenarios C and D. In the dataset used, no cities had the rural designation, and the classifications of each city are shown in Table 2-21.

Then the frequencies of cleaning were average across each classification bin to determine the average number of days between cleaning. These averages were rounded to regular frequencies. This resulted in a frequency of once a month in urban areas and six times per year in suburban areas. In the absence of any rural information, a cleaning frequency of two times a year, which is the lowest frequency reported in the survey, was selected for these locations. It was assumed that street cleaning has a 100% efficiency in removing wheel weights such that the entire reservoir of wheel weights along the curb is eliminated after each street cleaning event.

Table 2-21. Street Cleaning Statistics and City Classifications

City	State	Arterial	Central Business District	Residential	Population	Population Density	Classification

Table 2-21. Street Cleaning Statistics and City Classifications

City	State	Arterial	Central Business District	Residential	Population	Population Density	Classification
Oakland	CA	Daily		Biweekly	397,067	7,126	Urban Area
San Diego	CA		Weekly	Monthly	1,256,951	3,772	Urban Area
San Leandro	CA			Monthly	78,030	6,051	Urban Area
Long Beach	CA	Weekly	Weekly	Weekly	472,494	9,150	Urban Area
Mountain View	CA			Biweekly	70,090	5,863	Urban Area
San Jose	CA	Biweekly	Biweekly	Monthly	929,936	5,118	Urban Area
La Mesa	CA	2x/week	2x/week	Monthly	53,043	5,912	Urban Area
Sunnyvale	CA			Monthly	130,519	6,006	Urban Area
Union City	CA	Biweekly	Biweekly	Biweekly	69,477	3,474	Urban Area
Danville	CA	Monthly	Monthly	Monthly	41,540	2,306	Urban Cluster
Dublin	CA		Weekly	Biweekly	41,840	2,381	Urban Cluster
Elk Grove	CA	Monthly		3x/year	129,184	No data	Urban Area
Santee	CA	Weekly	Weekly	Biweekly	52,530	3,299	Urban Area
Greeley	CO	Biweekly	Weekly	5x/year	89,046	2,573	Urban Area
Fort Collins	CO		2x/week	2x/year	129,467	2,550	Urban Area
Denver	CO		Biweekly	8x/year	566,974	3,617	Urban Area
Thornton	CO	Biweekly		1x/year	109,155	3,067	Urban Area
Arvada	CO	6x-7x/year	6x – 7x/year	6x-7x/year	104,830	3,128	Urban Area
Tampa	FL	Weekly	Weekly	6x/year	332,888	2,708	Urban Area
Gainsville	FL	Monthly	2x/week	9x/year	108,655	1,981	Urban Area
Urbandale	IA	3x/year	3x/year	3x/year	37,173	1,405	Urban Cluster
Iowa City	IA	Monthly	Weekly	Monthly	62,649	2,575	Urban Area
Sioux City	IA	5x/year	5x/year	5x/year	83,262	1,551	Urban Area
Overland Park	KS	7x/year	Monthly	3x/year	166,722	2,627	Urban Area
Hanover Park	IL	8x/year	8x/year	8x/year	37,161	5,637	Urban Cluster
Evanston	IL	Biweekly		4x/year	75,543	9,579	Urban Area
Elgin	IL	Biweekly	2x/week	6x/year	101,903	3,780	Urban Area
Burr Ridge	IL	9x/year	9x/year	9x/year	10,408		Urban Cluster
Champaign	IL		Daily	8x/year	73,685	3,974	Urban Area
.Fort Wayne	IN	Biweekly	Weekly	4x/year	248,637	2,606	Urban Area
Cambridge	MA	Biweekly		9x/year	101,365	15,763	Urban Area
Salem	MA			9x/year	41,343	4,989	Urban Cluster
Saco	ME	Biweekly		9x/year	16,822		Urban Cluster
Kansas City	MO	4x/year	Weekly	4x/year	447,306	1,408	Urban Area
St. Joseph	MO	2x/year	2x/year	2x/year	72,651	1,688	Urban Area
Great Falls	MT	Biweekly	Daily	4x/year	56,215	2,909	Urban Area
Lincoln	NE			3x/year	241,167	3,022	Urban Area
Manchester	NH	Monthly	2x/week	3x/year	109,497	3,242	Urban Area
Albuquerque	NM	Biweekly	2x/week	Biweekly	504,949	2,483	Urban Area
Rochester	NY	2x/week	Daily	Biweekly	208,123	6,134	Urban Area
Albany	NY	Weekly	Weekly	Weekly	93,963	4,474	Urban Area
Toledo	OH	9x/year	2x/week	9x/year	298,446	3,890	Urban Area
Fairfield	OH	Biweekly	Weekly	5x/year	42,248	2,006	Urban Cluster

Table 2-21. Street Cleaning Statistics and City Classifications

City	State	Arterial	Central Business District	Residential	Population	Population Density	Classification
Macedonia	OH	2x/year	2x/year	2x/year	9,224		Urban Cluster
Marysville	OH	Weekly	Weekly	Monthly	18,212		Urban Cluster
Tulsa	OK	8x/year		4x/year	382,872	2,152	Urban Area
Albany	OR	Biweekly	Weekly	Monthly	46,213	2,573	Urban Cluster
Eugene	OR	Weekly	2x/week	Monthly	146,356	3,403	Urban Area
Pittsburg	PA	Weekly	2x/week	2-4x/year	312,819	6,020	Urban Area
Town of Lower Marion	PA	3x/year		3x/year	59,850		Urban Area
Knoxville	TN		Weekly	Monthly	182,337	1,876.60	Urban Area
San Antonio	TX	4x/year		2x/year	1,296,682	2,809	Urban Area
Dallas	TX	Monthly	Daily	None	1,232,940	3,470	Urban Area
El Paso	TX	Biweekly	Daily	4x/year	609,415	2,263	Urban Area
Austin	TX		Daily	6x/year	709,893	2,610	Urban Area
Ogden	UT	3x/year	3x/year	3x/year	78,086	2,898.90	Urban Area
Hampton	VA	Monthly		Monthly	145,017	2,828	Urban Area
Janesville	WI		5x/year	4x/year	62,998	2,160	Urban Area
Eau Claire	WI	3x/year	3x/year	3x/year	63,297	2,037.80	Urban Area
Milwaukee	WI		Weekly	Monthly	573,358	6,215	Urban Area

1.2.5 Fraction of Wheel Weights Gathered by Hobbyists Each Day

Hobbyists are known to gather wheel weights from along the roadway, thereby contributing another loss process. Ignoring the impact of hobbyist collectors may tend to overestimate the risks from lead wheel weights. However, there is no data available to inform the decision of the fraction removed by hobbyists. the fraction of lead wheel weights gathered by hobbyists per day was set at zero (0) to represent a conservative estimate.

1.2.6 Approach for Estimating Roadway Lead Dust Emissions

The most general method of modeling roadway lead dust emissions would be by tracking the mass of intact wheel weights, the mass of lead dust on the roadway, and the mass of roadway dust emitted each day. While the creation of such a model is not overly complex, the computations are considerably simplified if steady state conditions are assumed. At steady state, the mass of lead dust, the mass of intact lead wheel weights, the mass of silt, and the emission rate of lead dust are all constant. The Root (2000) study observes that steady state conditions are rapidly achieved on a roadway; empirical calculations made for this project also support this conclusion, with steady state conditions typically being reached within one year.

Lead dust emissions have therefore been computed at “average” steady state conditions. In order to avoid excessive conservatism and unrealistic results, it was assumed that street cleaning occurs at regular intervals. The mathematical relationship between the fall off rate of intact wheel weights and the average lead dust emission rate at steady state conditions was derived as follows:

Let:

F = the fall off rate of intact lead wheel weights from cars onto the roadway (in kg per day)

X = the mass of intact lead wheel weights on the roadway (in kg)

Y = the mass of lead dust (originating from the pulverization o f lead wheel weights)on the roadway (in kg)

p = pulverization rate (the fraction of lead wheel weights that are converted to lead dust per day)

u= street cleaning rate (the fraction of lead wheel weights that are removed from the road per day)

e = emission rate (the fraction of roadway lead dust that is suspended into the air by vehicles per day)

Mass balance considerations dictate that:

4. the change in mass of lead wheel weights on the roadway on a given day will equal the mass of lead wheel weights falling off from cars onto the roadway that day less the mass of wheel weights pulverized to dust on the roadway that day less the mass of wheel weights removed by road cleaning that day (see Figure 1); and

5. the change in mass of lead dust on the roadway on a given day will equal the mass of lead dust added to the roadway that day by pulverization less the mass of lead dust suspended into the air by passing automobiles on that day.

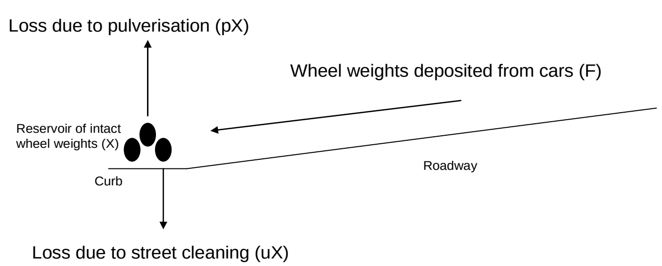


Figure 1. Diagram of the processes governing the stock of wheel weights in the curb

Using the symbols defined above, these mass balance equations may be expressed mathematically in terms of the following differential equations:

$$\frac{\square\square}{\square\square} = \square - \square\square - \square\square \qquad (1)$$

and

$$\frac{\square\square}{\square\square} = \square\square - \square\square \qquad (2)$$

At steady state, the mass of intact wheel weights (X) and the mass of lead dust (Y) are constant, implying that $\frac{\square\square}{\square\square} = 0$ and $\frac{\square\square}{\square\square} = 0$.

Setting $\frac{\square\square}{\square\square} = 0$ and $\frac{\square\square}{\square\square} = 0$ in equations (1) and (2) above, results, after some algebraic manipulation, in the following steady state relationships:

$$\square_{\square\square} = \frac{\square}{(\square + \square)} \qquad (3)$$

and

$$\square\square_{\square\square} = \square\frac{\square}{\square + \square} \qquad (4)$$

Equation (4) illustrates how the steady state emission of lead dust in the air from the roadway is a fraction of the fall off rate of intact lead wheel weights onto the roadway. If street cleaning does not exist, then at steady state the emission of lead dust equals wheel weight deposition on the roadway.

A complication that prevents a purely analytic estimation of the steady state emission rate of lead dust is that the street cleaning rate u in the equation above is not a constant but varies with time (to reflect the reality that street cleaning occurs not daily but at a periodic frequency). For a street with a monthly cleaning frequency, it was assumed that u would equal zero for days 1-29 and then equal 1 on the thirtieth day, after which it would assume the value zero for the next 29 days, and so on. This assumes that street cleaning removes the entire stock of wheel weights on the curb on the days that it occurs. Consequently, the average steady state emission rate was estimated empirically using a dynamic spreadsheet model that directly simulates equation 1 above. The steady-state assumption for equation 2 still holds so that the amount of lead pulverized each day equals the amount emitted in dust each day.

The occurrence of cyclical street cleaning prevents the realization of a true unvarying steady state; instead a “cyclical steady state” is achieved in which the emission rate and other variables repeat the same values on a cyclical basis related to the cleaning frequency. Figure 8 shows the wheel weight deposition rate as well as the cyclical dust emission rate. For the purposes of computing average exposure and risk, the average dust emission rate across the cycles was used.

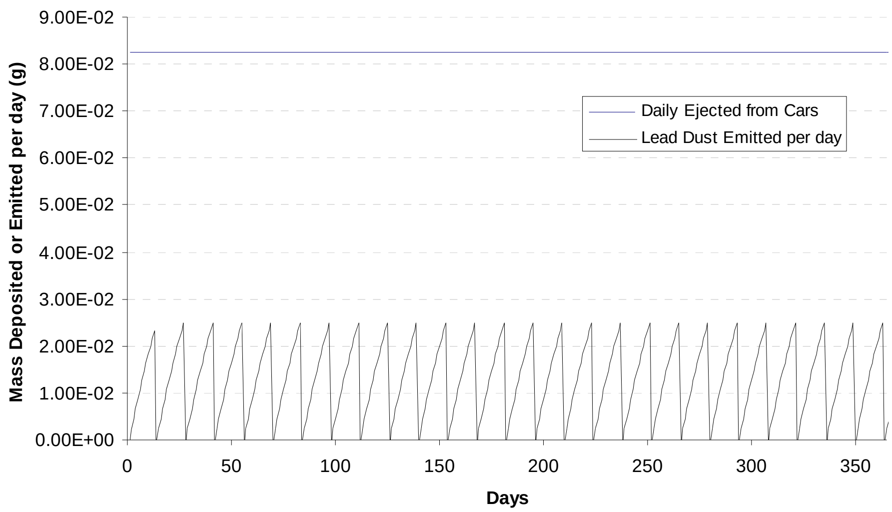


Figure 8. Mass of Wheel Weights Emitted Per Day in a 1 m Urban Segment of Road and the Cyclical Lead Dust Emitted from the Roadway Each Day

Table 2-22 presents empirically computed ratios of average steady state roadway lead dust emission rates to lead wheel weight fall out rates for three cleaning frequency scenarios. It is apparent that as the frequency of street cleaning decreases the emission rate increases. This reflects the reality that the longer wheel weights remain on the roadway, the more they will be pulverized, and the more lead dust emissions will increase for a given ejection rate. As the pulverization fraction increases, emission rates will rise too, reflecting the increased stock of lead dust on the road. As a conservative assumption, the pulverization rate determined in the Root study has been used in all scenarios.

Table 2-22. Roadway Lead Dust Emission Rates in Urban, Suburban and Rural Areas

Scenario	p (Pulverization Fraction per day)	Cleaning Frequency (in days)	(Ratio of Average Steady State Emission Rate of Lead Dust to Fall Off Rate of Wheel Weights)
Urban	0.0272	14	0.16
Suburban	0.0272	60	0.53
Rural	0.0272	183	0.81

Water run off of roadway lead dust has not been modeled, although this may be a significant loss process. Furthermore, in areas which receive snowfall, snow ploughs may remove wheel weights and lead dust from off the roadways. This too has not been considered. The omission of these loss processes increase the conservatism of the results.

To estimate the final area source emission rates, the wheel weight ejection rate (1.56 E-6 kg/VMT) was multiplied by the ratio of the steady state emission to the fall off rate assuming street cleaning, as shown in the final column of Table 2-22. Then, this emission rate was multiplied by the vehicle counts on the individual roads (high volume and low volume) in the domain as shown in Table 2-13 to get the total mass emitted per day. These values were divided by the area of the road section, the masses were converted to micrograms, and the daily emissions were divided by 86,400 seconds per day to determine the emission flux in units of $\mu g/m^2/s$. Finally, the emissions were normalized by a factor of 1E8 to allow increased modeling precision. This factor is then divided out when calculating the modeled air concentrations and depositions at the maximally exposed home.

1.3 Yard SOIL module

1.3.1 Depth of Surface Soil Compartment

The thickness of the surface soil layer assumed in TRIM.FaTE model simulations performed for EPA OAQPS ranges from 1 cm for non-agricultural soils to 20 cm for tilled agricultural soils (USEPA, 2009b). Although yard soils are not expected to be tilled, they may be mowed, raked, landscaped, or used for gardening. In order to be adequately conservative, a yard surface soil layer thickness of 1 cm was assumed.

1.3.2 Density and Porosity of Soil Particles

The typical soil particle density assumed in TRIM.FaTE model simulations performed for EPA OAQPS is 2,600 kg/m^3. This value is taken from McKone et al. (2001). In addition, McKone et al. also provided an estimate for the soil porosity of 20% or 0.2.

1.3.3 Residence Time of Lead in Surface Soil

A literature search was conducted to estimate the residence time of lead in surface soil. The following studies were reviewed: Tyler (1978), Miller and Friedland (1994), Erel (1998), USEPA (2001), Kaste et al. (2003), Semali et al. (2004), Kaste et al. (2005), Klaminder et al. (2006a), Klaminder et al. (2006b), and Miretzky and Fernandez-Cirelli (2007), as shown in Table 2-23.

There were a number of variations in each of the studies reviewed. Studies were conducted in different areas of the world, including the Northeastern United States, Israel, Sweden, and France. Studies derived the residence time using a number of different methods, including experimental measurement of lead through soil, mass-balanced source models, tracer isotope tracking within soil, or chronosequencing lead in soil gradients. In addition, results were presented in numerous formats including residence times, response times, half lives, and 10% removal times. All half-life and 10% removal calculations were converted to response time, and calculations were made to ensure all definitions in the papers of residence time and response time were equivalent to each other.

Table 2-23. Lead in Soil Residence Time Literature Search Results

Paper	Year	Reported Time (yrs)	Residence Time (yrs)	Location
Tyler (1978)	1978	700-900 (10% removal)	6650-8550	Forest in Sweden
Miller and Friedland (1994)	1994	17-77 (response)	37-167	Northeast US
Erel (1998)	1998	100-200 (residence)	100-200	Israel, farmland and forest
USEPA (2001)	2001	1000 (half life)	1442	Unknown
Kaste et al. (2003)	2003	60-150 (response)	60-150	Northeast US
Semali et al. (2004)	2004	700 (half life)	1000	France
Kaste et al. (2005)	2005	50-150 (response)	50-150	Northeast US
Klaminder et al. (2006a)	2006	150 (residence)	150	Forest in Sweden
Klaminder et al. (2006b)	2006	50-250 (residence)	50-250	Forest in Sweden
Mireztky and Fernandez-Cirelli (2007)	2007	740-5900 (half life)	1070-8500	Unknown

A number of factors effect the residence time of lead in the soil. The carbon flux within the soil layer is closely correlated to the residence time of lead. In newer growth forests, residence times are smaller than older growth forests. There is greater turnover of carbon in these newer growth forests. Older growth forests may have a higher organic carbon content in the upper layers or soil, but it may be broken down more slowly (Klaminder et al., 2006b). In addition, warmer climates may have quicker turnover of carbon and thus shorter lead residence times (Miller and Friedland, 1994).

Overall, the values reported in the studies vary over a wide range. For the yard soil module, a value of 1,000 years was selected. This value is in the upper range of literature values and represents a moderately conservative estimate of the residence time.

1.3.4 Background Home Yard Lead Soil Concentration

Background home yard lead soil concentrations were determined for the hypothetical model locations using proxy locations for each type, as shown in Table 2-24. For the urban location, a high soil concentration was used. The value was taken from a study of the concentrations in yards in Dorchester, MA (Hynes, Maxfield et al., 2001). The selected value represents the arithmetic mean of lead in surface soil in the North Dorchester Project area.

For the rural location, both a high and low soil concentration area are modeled. For the high soil concentration yard, values from a study measuring soil concentrations in residential Minnesota were used (Schmitt, Trippler, et al. 1988). The value represents the maximum value for the front yard lead concentrations for the "outstate" classification. For the low soil concentration area, values from a study measuring lead concentration in rural topsoil in South Carolina were used (Aelion, Davis, et al., 2008). The value represents the mean lead concentration in the less contaminated strip of land from the study (strip 1).

For the suburban location, a low soil concentration area is modeled. The Schmitt et al. study mentioned above for rural locations was used, and the selected value represents the median front yard lead concentrations for the "outstate" classification.

Table 2-24. Background Home Yard Lead Soil Concentration

Urban, High Soil Concentration (Scenarios A and B)	Rural, High Soil Concentration (Scenario C)	Rural, Low Soil Concentration (Scenario D)	Suburban, Low Soil Concentration (Scenario E)
1463 µg/g	656 µg/g	12 µg/g	37 µg/g

1.3.5 Approach for Estimating the Yard Soil Concentration From Wheel Weights

To estimate the contribution to the yard soil concentration from the wheel weight lead emission, a vertical mass balance is used.

Let:

M = the mass of lead in the soil (in µg)

C = the concentration of lead in the soil (in µg/g)

D = the deposition rate of lead into the soil (in $\mu g/m^2/year$)

τ = the residence time of lead in soil (in years)

ρ = the density of the soil (g/m^3)

ϕ = the porosity of the soil (fraction)

A = the area of the yard (m^2)

d= the depth of the top soil layer (m)

Mass balance considerations dictate that:

> The change in the mass of lead in the soil equals the deposition input from above less the loss due to vertical colloidal transport.

Using the symbols defined above, this mass balance equation may be expressed mathematically in terms of the following differential equation:

$$\frac{dM}{dt} = D \times A - \frac{M}{\tau} \qquad (5)$$

This equation assumes that the colloidal transport can be captured by first order removal with a rate constant equal to $1/\tau$ (which is equivalent to the residence time). At steady state, the mass of lead in the soil is not changing, so

$$\frac{M}{\tau} = D \times A \qquad (6)$$

The mass of lead in the soil can be converted to concentration in units of mass of lead per mass of soil by using the soil density, porosity, and soil thickness,

$$C = \frac{D \times \tau}{d \times \rho \times (1 - \phi)} \qquad (7)$$

Thus, given the total deposition of lead in the yard from the AERMOD model, the residence time in the soil, the soil depth, the soil density, and the porosity, the lead concentration due to wheel weights can be calculated using equation 7.

1.4 Indoor air/dust module

1.4.1 Background Ambient Air Concentration

The background ambient air concentration was calculated using air monitoring information from the EPA's Air Quality System (AQS; USEPA, 2010a) DataMart database. Average annual concentrations from all monitoring locations in the AQS system measuring lead total suspended particulate (TSP) at standard temperature and pressure (STP), or parameter ID 12128. Data from 2008 were used, since in 2009 monitors began using updated reporting methods due to the most recent Pb NAAQS rules; however, because different monitors used different reporting methods, the statistical strength of averaging for any one reporting type was greatly diminished.

The AQS database includes a field named “Monitoring Objective” that specifies the reason that a monitor was placed in each location. Monitors labeled “source oriented”, “quality assurance” (duplicate monitors at the same site, which may bias results), or “Unknown” were removed from the analysis, as it is likely that the results from these sites will bias background ambient air concentrations. In addition, numerous monitors were located in the town of Herculaneum, Missouri which operates the largest lead smelter in the United States. All sites located in Herculaneum were also removed, regardless of the stated monitoring objective.

Monitoring stations were assigned to rural, suburban, or urban locations in AQS using the “Location” field. If the location was unknown, the latitude and longitude was viewed in Google Earth® and an assignment was made by professional judgment. Only locations with residential and commercial land use types were included.

The remaining monitors’ annual average concentrations in µg/m^3 in each station type (rural, suburban, or urban) were used to give estimates of the average, standard deviation, and median ambient air concentrations in each location, as shown in Table 2-25. The average concentrations were selected for use in the modeling framework.

Table 2-25. Ambient Air Concentrations from the AQS Monitoring Network

Description	N	Average (µg/m^3)	Standard Deviation (µg/m^3)	Median (µg/m^3)
Urban and City Center	31	0.025	0.054	0.0075
Rural	8	0.011	0.006	0.0130
Suburban	39	0.014	0.022	0.0067

1.4.2 Penetration Fraction of Ambient Air Into Home

The penetration fraction captures the ratio of the indoor concentration from outside sources to the ambient (outdoor) concentration. The penetration fraction was set equal to 1.0, taken from Thatcher and Layton (1995). The paper reported penetration for lead-containing particles in a home in California, and the penetration fraction was near one for all size classes.

1.4.3 Dust Regression Equation

The concentration of lead in indoor dust inside a home is determined by the outdoor soil concentration tracked into the home, the indoor lead paint concentration in the home, the ambient air concentration, the cleaning frequency, the occupancy level, and the nature of non-lead particulate sources in the home. Lead wheel weights will contribute lead mass to the outdoor soil concentration and ambient air concentration, which will in turn affect the indoor lead dust concentration. In addition, different housing vintages in the different scenarios will have different levels of lead in the interior paint. The National Survey of Lead-Based Paint in Housing ("HUD Survey Data", USEPA, 1995) provides information on the lead dust concentration determined from particulate collected using Blue Nozzle vacuum samplers, yard-wide average lead soil concentrations, the maximum observed indoor XRF lead paint concentrations, and the housing vintage for 312 homes. These data are used to determine a regression equation relating the interior dust concentration with the outdoor soil concentration and the paint concentration. The ambient air concentrations were not captured in the survey, so these values could not be included in the regression equation.

Using Statistica®, a multiple linear regression equation was developed relating the indoor dust concentration to the outdoor soil concentration and indoor paint concentration. Both the untransformed and the natural-log-transformed variables were used in order to determine which linear regression captured the largest portion of the observed variance. Statistics from the two different fits are shown below in Table 2-26. The regression based on the untransformed variables captured little of the total variance and did not indicate significance at the p=0.01 level. Thus, the regression based on the natural-log-transformed variables was selected. This regression has an adjusted R^2 of 0.24, representing reasonable predictive power but indicating much of the variance is explained by other factors not included in the regression or captured in the survey, such as those mentioned above (ambient air concentration, cleaning frequency, occupancy level, etc.). The equation for the indoor dust concentration in □g/g becomes

$$\text{Dust} = 44.3 \times \text{Soil}^{0.33} \times \text{Paint}^{0.22}$$

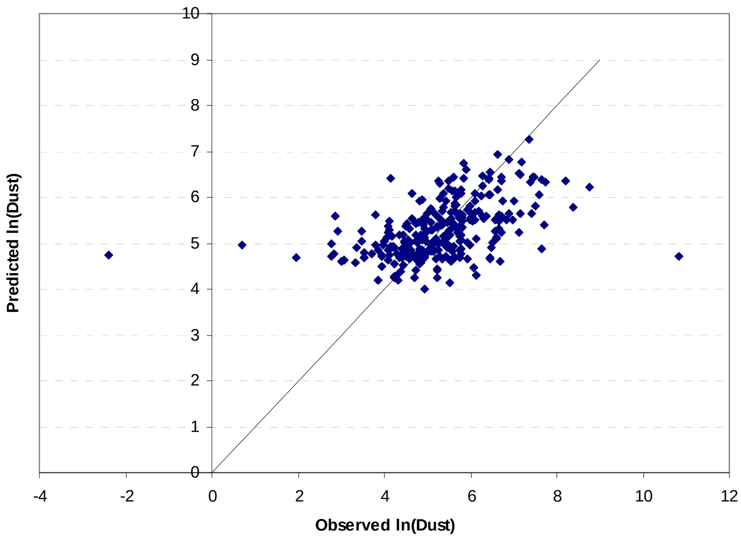
where *Soil* is the concentration in the soil in □g/g and *Paint* is the concentration of lead in the interior paint in mg/cm^2. Figure 9 shows the predicted ln(dust) as a function of the observed ln(dust), where the solid line denotes a 1:1 correspondence.

Table 2-26. Statistics of the Multiple Linear Regression for Dust Concentration

Transformation	Variable	Beta	Standard Error of Beta	B	Standard Error of B	T(294)	p level
Untransformed	Intercept			445.71	203.60	2.19	0.029
	soil	0.0384	0.061	0.18	0.2803	0.63	0.530
	XRF	0.0181	0.061	17.39	58.79	0.30	0.768
Natural-log- transformed	Intercept			3.79	0.2220	17.08	< 1e-10
	ln(soil)	0.3926	0.057	0.33	0.0483	6.85	< 1e-10
	ln(XRF)	0.1732	0.057	0.22	0.0729	3.02	0.003

	R	R^2	Adjusted R^2	F	P level	Standard Error
Untransformed	0.047	0.0022	--	0.33	< 0.72	2961.8
Natural-log-transformed	0.5	0.25	0.24	48.5	< 1e-10	1.08

Figure 9. Predicted ln(Dust) as a Function of the Observed ln(Dust)



Paint concentrations are not explicitly considered in the overall wheel weight modeling approach. However, the housing vintage in each scenario has been specified. Thus, the average paint concentration across all homes in the HUD Survey in each specified vintage bin was calculated and plugged into the dust equation to create vintage-specific equations, as shown in Table 2-27 below.

Table 2-27. Dust Regression Equation By Housing Vintage

	Pre 1940 Vintage (Scenarios A and C)	Post 1980 Vintage (Scenarios B, D, and E)
Average XRF Paint Concentration (mg/cm^2)	3.69	0.519
Dust Equation	$\text{Dust} = 59.0 \times \text{Soil}^{0.33}$	$\text{Dust} = 38.3 \times \text{Soil}^{0.33}$

1.5 time exposure module

The time exposure module takes into account the time the child spends outside the home and the exposure concentrations during those times. For this assessment, it is assumed that if a child is outside the home they are at a nearby school or daycare facility. Because these facilities are expected to be close to the child's home, the assumption was made that the exposure concentrations during the time outside the home are equal to the exposure concentrations in the home.

1.6 blood lead (ieubk) module

The blood lead model requires a number of inputs aside from the air, soil, and dust lead concentrations. Table 2-28 shows the inputs and the proposed values for each. As a starting point, the values were set to those used in the development of the current lead NAAQS level (USEPA, 2007) and in the lead renovation and repair rule (USEPA, 2008c). Then, several input values were updated with data from more recently published literature. Several other values, however, were found to still reflect the best information available. These included water lead concentration, lead absorption fractions, dietary lead intake, and the fraction of ingested soil+dust which is soil.

In 2008, the U.S. EPA published a new edition of its Child-Specific Exposure Factors Handbook, from which updated mean values for total indoor/outdoor dust ingestion, water consumption, and ventilation rate were derived (USEPA, 2008b). Where ages were expressed as a range in that report, rates for intermediate ages were interpolated using linear trendlines.

The IEUBK value for maternal blood Pb level was updated using data from the most recent NHANES survey. These data from 2007 and 2008 reveal that the GM blood lead level among women aged 18 through 45 is 0.847 µg/dL. This was computed using the NHANES laboratory sample data and included nationally-representative sample weights (CDC, 2009).

Table 2-28. Blood Pb Model Input Values

Group	Parameter	Parameter Name	Parameter Value IEUBK Default Age Ranges (Years) 0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	Basis/Derivation
Inhalation	Daily ventilation rate (cubic meters [m^3]/day)	Ventilation rate	5.4	8.0	9.5	10.9	10.9	10.9	12.4	USEPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages
	Absolute inhalation absorption fraction (unitless)	Lung absorption	0.42							USEPA (1989)
	Indoor air Pb concentration	Indoor air Pb concentration (percentage of outdoor)	100%							These values are taken directly into account when developing the exposure concentrations
	Time spent outdoors	Time spend outdoors (hours/day)	Not used							
Drinking Water Ingestion	Water consumption (L/day)	Water consumption (L/day)	0.36	0.271	0.317	0.349	0.380	0.397	0.414	USEPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages
	Water Pb concentration (µg/L)	Lead concentration in drinking water (µg/L)	4.61							GM of values reported in studies of United States and Canadian populations (residential water) as cited in USEPA (2006), section 3.3 Table 3-10), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA, 2008c)
	Absolute absorption (unitless)	Total percent accessible (IEUBK)	50 % (Single value used across all age ranges)							Assumed similar to dietary absorption (see "Total percent accessible" under Diet below), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA, 2008c)
Diet	Dietary Pb intake (µg/day)	Dietary Pb intake (µg/day)	3.16	2.6	2.87	2.74	2.61	2.74	2.99	Estimates based on the following: (1) Pb food residue data from U.S. Food and Drug Administration (U.S. FDA) Total Diet Study (USFDA, 2001), and (2) food consumption data from NHANES III (CDC, 1997), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA,

Table 2-28. Blood Pb Model Input Values

Group	Parameter	Parameter Name	Parameter Value: IEUBK Default Age Ranges (Years) 0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	Basis/Derivation
										2008c)
	Absolute absorption (unitless)	Total percent accessible	50%							Alexander et al. (1974) and Ziegler et al. (1978) as cited in USEPA (2006, section 4.2.1), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA, 2008c)
Outdoor Soil/Dust and Indoor Dust Ingestion	Outdoor soil/dust and indoor dust weighting factor (unitless)	Outdoor soil/dust and indoor dust ingestion weighting factor (percent outdoor soil/dust)	45%							This is the percent of total ingestion that is outdoor soil/dust. Value reflects best judgment and consideration (results published by van Wijnen et al. (1990), as cited in (USEPA, 1989), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA, 2008c)
	Total indoor dust + outdoor soil/dust ingestion (mg/day)	Amount of outdoor soil/dust and indoor dust ingested daily (mg)	60	110	110	110	110	110	110	USEPA Child-Specific Exposure Factors Handbook (2008b), excluding cases of soil-pica and geophagy
	Absolute gastrointestinal absorption (outdoor soil/dust and indoor dust) (unitless)	Total percent accessible (IEUBK)	0.30 for both outdoor soil/dust and indoor dust							Reflects evidence that Pb in indoor dust and outdoor soil/dust is as accessible as dietary Pb and that indoor dust and outdoor soil/dust ingestion may occur away from mealtimes (USEPA 1989), as in the Lead NAAQS (USEPA, 2007) and Lead Renovation, Repair, and Painting Rule (USEPA, 2008c)
Other	Maternal PbB (µg/dL)	Maternal PbB concentration at childbirth, µg/dL	0.847							NHANES 2007-2008, national weighted GM of all women aged 18-45 (CDC, 2009)

1.7 iq module

Lanphear et al. (2005) derived regression relationships between several blood Pb metrics (lifetime averages and measurements made concurrently with the IQ test administration) and IQ test results based on linear, cubic spline, log-linear, and piecewise linear equations. Similar to the EPA Lead Renovation, Repair, and Painting (RRP) Rule analysis (USEPA, 2008c), the regression using piecewise linear equations and the lifetime blood lead average was selected to analyze the lead wheel weights IQ changes. The model has a blood lead "cutpoint" at 10 □g/dL where the slope of the concentration-response curve goes from a more steep slope at low blood lead levels to a less steep slope at higher blood lead levels. The equation relating blood lead to the change in IQ is then:

$PbB < 1$ $\quad IQ\ change = 0$

$PbB = 1 \text{ to } 10$ $\quad IQ\ change = PbB * -0.88$

$PbB > 10$ $\quad IQ\ change = -8.8 + (PbB - 10) * -0.10$

where:

PbB = Lifetime average of the blood lead level

As shown in the above equations, no IQ changes are predicted for blood lead concentrations less than 1.0 µg/dL. This assumption was made in recognition of the lack of data in this blood lead range in the Lanphear et al. (2005) study cohorts.

The modeling framework for the near-roadway residence includes resuspension of road dust into the air and the subsequent dispersion and deposition of this lead-containing dust into nearby yards. However, the model does not include the resuspension of contaminated yard soil into the air. In order to include this process, a full multi-media model which simultaneously models both air and soil processes would have to be used; however, these models tend to have less sophisticated dispersion algorithms than the air-only AERMOD model.

To determine the possible uncertainty associated with excluding yard soil resuspension, a literature search was conducted. The search focused on peer-reviewed journal articles which address resuspension of lead or other metals from soil to the air. The search was conducted using Google Scholar® and using search combinations of the following words: "lead", "metal", "resuspension", and "reentrainment". Table A.3 in Appendix A lists the papers found in the literature search.

In general, the papers suggest that resuspension of contaminated soil can be a large contributor to ambient air concentrations. Harris and Davidson (2009) employ a mass balance model to conclude that sources of lead due to the resuspension of contaminated soil/dust are a factor of ten higher than direct sources of lead in the South Coast Air Basin in California. They cite the main contributor of lead in the soil to be from historical deposition in the era of leaded gasoline, and the current sources due to resuspension include both yard soil and roadway soil. Sabin et al. (2006), however, found that much of the airborne lead in Los Angeles was due to resuspension from roadways, and concentrations of lead in air returned to near-background levels within 10 to 150 m of the roadway. Hosiokangas et al. (2004) also found that roadways were a major contributor to airborne lead levels (27%) in Finland, and the windspeed tended to be the major determinant of how much lead was resuspended. These papers suggest that resuspension of contaminated soil/dust is a major contributor to airborne lead, but much of this resuspension occurs on roadways where car turbulence creates an effective mechanism for suspending the dust.

Thus, excluding yard resuspension will tend to underpredict the yard air lead concentrations; however, the dominant source to a yard next to a roadway is likely the resuspended roadway lead rather than lead resuspended from the yard itself. Tithe exclusion of yard resuspension remains a recognized limitation of the modeling approach.

3.1 media concentrations

3.1.1 Scenarios A and B

The concentrations in the receptor yards relative to the high and low volume streets for the urban scenario 3km grid are shown in Figure 10. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic and is indicated with a star. At this point, the concentration is 0.017 $\mu g/m^3$, and the total deposition (wet and dry) is 0.0011 g/m^2/year. The modeled concentration can be compared with the background concentration of 0.025 $\mu g/m^3$. In initial modeling efforts when street cleaning was not taken into account in the estimation of the lead emission rate, the modeled concentration was 0.054 $\mu g/m^3$, which is above the background concentration. However, the background concentration should include the contribution from wheel weights. This observation indicated the scenario was overly conservative and the cleaning frequency calculation was included to ensure more reasonable modeling results were achieved.

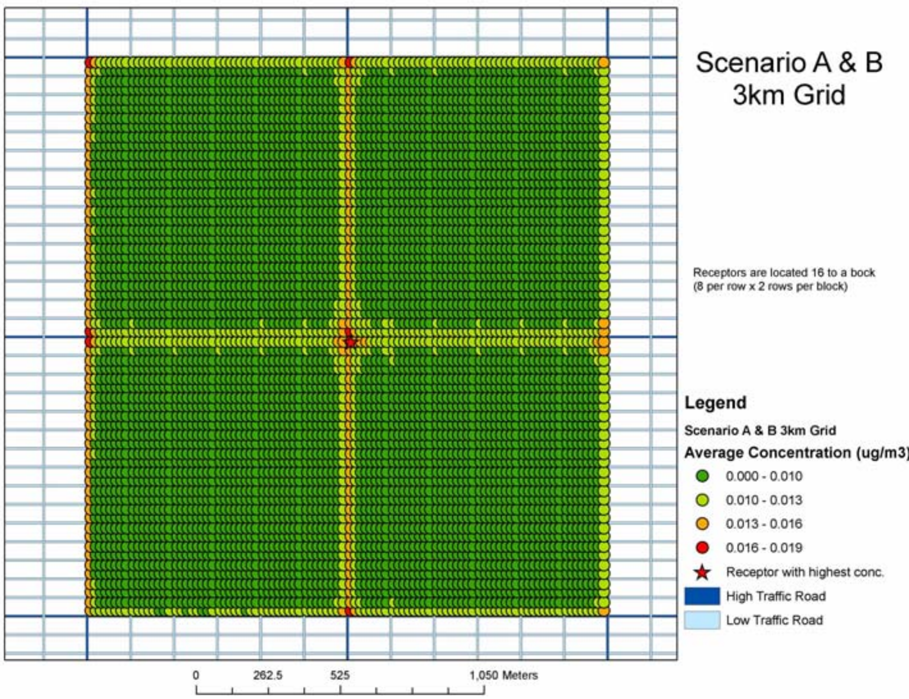


Figure 10. Modeled Concentrations in the Urban Scenario 3km Grid

3.1.2 Scenarios C and D

The concentrations in the receptor yards relative to the high and low volume streets for the urban scenario 3km grid are shown in Figure 11. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 7.8E-4 g/m^3, and the total deposition (wet and dry) is 5.3E-5 g/m^2/year. The modeled concentration can be compared with the background concentration of 0.010 g/m^3.

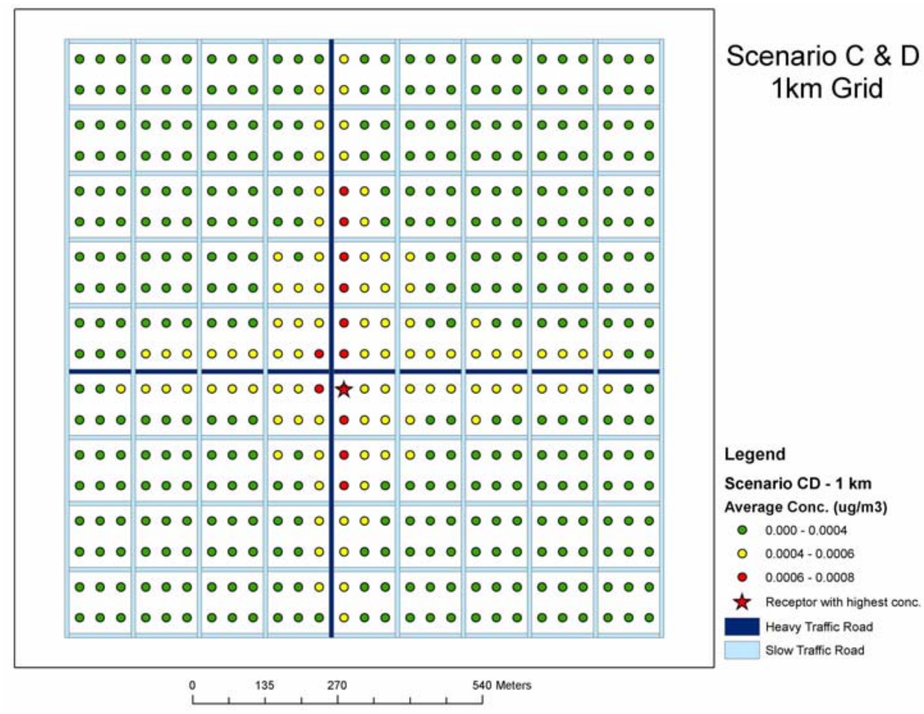


Figure 11. Modeled Concentrations in the Rural Scenario 1km Grid

3.1.3 Scenario E

The concentrations in the receptor yards relative to the high and low volume streets for the suburban scenario 2km grid are shown in Figure 12. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 2.1E-3 g/m^3, and the total deposition (wet and dry) is 1.4E-4 g/m^2/year. The modeled concentration can be compared with the background concentration of 0.014 g/m^3.

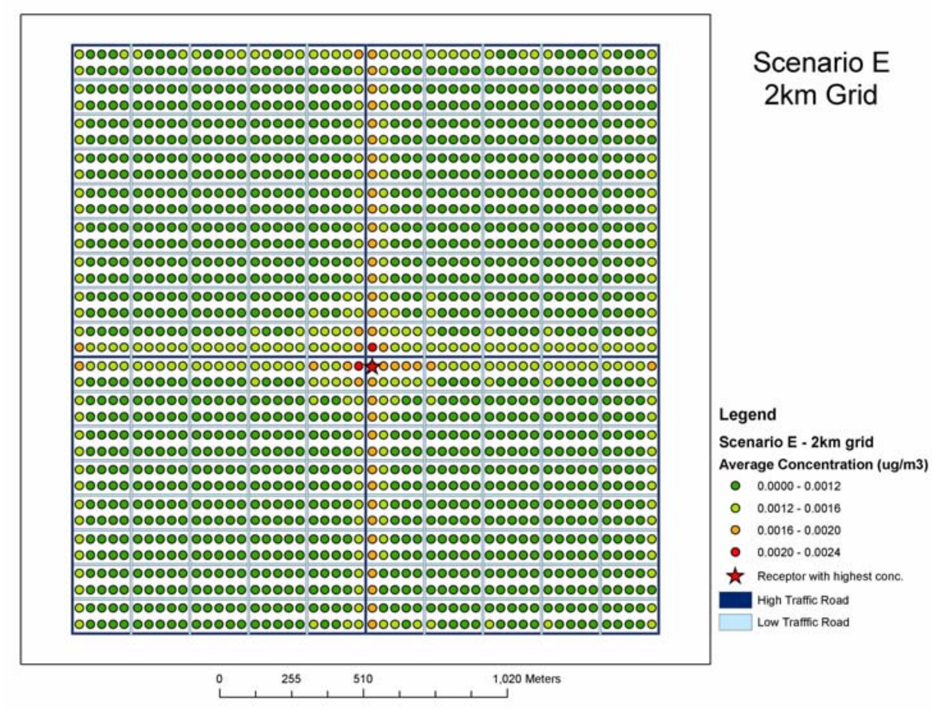


Figure 12. Modeled Concentrations in the Suburban Scenario 2km Grid

3.1.4 Summary of Media Concentrations in the Modeled Scenarios

In each scenario, the modeled air concentrations were binned into intervals which span the range of modeled concentrations in the domain. The bins were selected so that each scenario had three or four bins and the bin boundaries were equally-spaced. Then, the percentage of yards in each concentration bin was calculated using all the modeled yards on the eastern side of the grid. Because the wind is predominantly from the western direction, the eastern side of the grid has a larger contribution from upwind wheel weight emission and thus has a higher level of concentration precision than the western side of the grid. Table 0-1 shows the bin definitions and the percentage of eastern yards in each bin for the modeled scenarios.

Table 0-1. Modeled Air Concentration Bin Definitions

Model Scenario	Bin	Maximum Concen in Bin (µg/m^3)	Number of Modeled Yards in Bin In Eastern Portion of Domain	Proportion of Modeled Yards in Bin in Eastern Portion of Domain
Scenario A and B	Bin 1	0.0100	2543	85.9%
	Bin 2	0.0130	343	11.6%
	Bin 3	0.0160	70	2.4%
	Bin 4	0.0190	4	0.1%
Scenario C and D	Bin 1	0.0004	207	76.7%
	Bin 2	0.0006	53	19.6%
	Bin 3	0.0008	10	3.7%
Scenario E	Bin 1	0.0012	674	79.3%
	Bin 2	0.0016	135	15.9%
	Bin 3	0.0020	39	4.6%
	Bin 4	0.0024	2	0.2%

Next, the mean air concentration and deposition was calculated in each bin for each scenario. These concentrations were then used to also calculate the soil and dust concentrations corresponding to these mean concentrations. In addition, the maximum air concentration and deposition in the domain were used to find the media concentrations at the maximally exposed home. Table 0-2 shows these media concentrations calculated from the AERMOD modeling, the yard soil module, and the indoor dust module. The background estimates are presumed to include both the wheel weight and other lead source contributions. The wheel weight contribution in the table represents the portion of the total media concentration that is contributed by lead wheel weights. In the case of the dust concentration, this contribution is only approximate since the dust regression equation is nonlinear. The dust concentration was found using the 1) background soil concentration and 2) the background soil concentration minus the wheel weight contribution and then subtracting 2) from 1). In general, the wheel weight contributions are a small percentage of the total soil and dust concentrations, particularly in the high soil concentration and earlier housing vintage cases. The air concentration contribution is larger, varying from 8% in the rural case up to 70% in the urban case.

Table 0-2. Media Concentrations in the Modeled Scenarios

Scenario	Bin	Concentrations					
		Background Air (µg/m^3)	Wheel Weight Contribution to Air (µg/m^3)	Background Soil (µg/g)	Wheel Weight Contribution to Soil (µg/g)	Background Dust (µg/g)	Approximate Wheel Weight Contribution to Dust (µg/g)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	0.0250	0.0083	1463.0	25.0	658.5	3.7
	Bin 2 Mean		0.0112		35.0		5.3
	Bin 3 Mean		0.0142		44.8		6.7
	Bin 4 Mean		0.0169		54.7		8.3
	Max		0.0174		55.7		8.4
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin 1 Mean	0.0250	0.0083	1463.0	25.0	427.5	2.4
	Bin 2 Mean		0.0112		35.0		3.4
	Bin 3 Mean		0.0142		44.8		4.4
	Bin 4 Mean		0.0169		54.7		5.4
	Max		0.0174		55.7		5.5
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	0.0100	0.0003	656.0	0.9	504.9	0.2
	Bin 2 Mean		0.0005		1.4		0.4
	Bin 3 Mean		0.0007		2.1		0.5
	Max		0.0008		2.5		0.6
Scenario D: Rural area,	Bin 1 Mean	0.0100	0.0003	12.0	0.9	87.2	2.2

low soil lead concentration, post-1980 housing	Bin 2 Mean		0.0005		1.4		3.6
	Bin 3 Mean		0.0007		2.1		5.4
	Max		0.0008		2.5		6.6
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin 1 Mean	0.0140	0.0010	37.0	2.7	126.6	3.1
	Bin 2 Mean		0.0013		3.8		4.4
	Bin 3 Mean		0.0018		5.4		6.4
	Bin 4 Mean		0.0023		7.0		8.5
	Max		0.0023		7.1		8.7

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